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HYDROGEOLOGICAL INVESTIGATIONS
OF THE UPPER OTTAWA STREET
LANDFILL SITE
HAMILTON, ONTARIO

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THE UPPER OTTAWA STREET LANDFILL SITE
HAMILTON, ONTARIO


by

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PREFACE

The Upper Ottawa Street landfill, which occupies about 18 hectares, is located near the slope of the Niagara Escarpment in Hamilton, Ontario (Figure 1). A hydrogeological study of the site by Gartner Lee Associates (1979), done for the Regional Municipality of Hamilton-Wentworth, established that a water-table mound exists within the landfill and that the landfill is situated on permeable fractured bedrock consisting of dolomite, shale, siltstone and limestone. The landfill received municipal and industrial wastes during a period of more than two decades, beginning in the early 1950's and ending in 1980. Much of the industrial waste deposited in the landfill was in liquid form.

Hydrogeological investigations of the site by the University of Waterloo began in the fall of 1981 and continued until the end of 1983. The purpose of this investigation was to delineate the extent of migration of landfill-derived contaminants in the groundwater zone in the fractured rock and to determine the chemical composition of this zone. The design of this investigation was founded on the premise that contaminants from the landfill have entered the groundwater zone beneath the landfill and that they have been and are continuing to be transported along groundwater flow paths in networks of open fractures in the bedrock.

The topography of the land surface in the vicinity of the landfill is very irregular and therefore, it was expected that the water table would also be irregular in this area. The ambient groundwater in the bedrock has unusually high concentrations of natural organic compounds and of major inorganic ions. The network of open fractures in the bedrock is complex. Therefore, at the onset of the investigation it was expected that the pathways and rates of migration of landfill-derived contaminants would not be amenable to determination using conventional methods. A search of the literature provided no examples of detailed studies of contaminant migration at other landfill sites on bedrock with this degree of physical and chemical complexity.

In this investigation a combination of conventional and unconventional methods were used. A new device consisting of an assembly of several narrow-diameter piezometers for groundwater monitoring at several depth levels in single boreholes was developed in the earliest stage of the investigation. The design of the device was improved as the investigation proceeded. After this device was established as being practical and effective in the investigation of the Upper Ottawa Street Landfill site, it was used by other hydrogeologists at four sites elsewhere in Ontario, and at a site in New Brunswick.

In hydrogeologic studies of landfill sites on sand aquifers in Ontario, we have found that zones of contaminat-

ed groundwater are easily identified on the basis of chemical compounds that are very low or non-existent in the ambient groundwater but that are quite high in the contaminated groundwater zones downflow from the landfills. In contrast, it became apparent in the early stages of the investigation of the Upper Ottawa Street landfill site that the ambient groundwater at this site may contain nearly all of the compounds that at other sites are normally associated with landfill leachate. The strategy for the chemical studies of groundwater at this site, therefore, evolved into one much different than that which we have applied at other landfill sites in Ontario.

This report has two main parts and eight appendices. Part 1 provides a description of the geology, permeability and groundwater-flow conditions in the fractured bedrock at the site. Part 2 describes the results and interpretations of chemical analyses of water samples collected from the groundwater monitoring network.

Appendix A describes sampling procedures and protocols. The laboratory methods used for the chemical analyses are described in Appendix B. The results of the chemical analyses are listed in Appendices C, D and E.

The design, testing and use of the new device for multilevel groundwater monitoring in fractured rock is presented in Appendix F. This appendix is a paper that has been accepted for publication in the Canadian Geotechnical Jour-

nal in 1985. This device is currently undergoing minor design modifications by an Ontario company in Burlington that will soon be marketing the device in Canada and the United States.

The results and interpretations of detailed tests for hydraulic conductivity of the bedrock in a small part of the overall study area are presented in Appendix G. The main findings of these tests are also described, but only briefly, in Part 1 of this report.

Concurrent with the field investigations, a graduate student at the University of Waterloo developed a mathematical model for simulation of groundwater flow and contaminant transport in fractured media such as bedrock. This model was used to evaluate the influence of various characteristics of open fractures on contaminant migration in hypothetical systems. Although it is one of the most sophisticated models currently in existence, it was found that it could not be used to simulate actual contaminant migration at the Upper Ottawa Street Landfill site because the hydrogeologic conditions at this site are so complex. The available data on the geometry of the fractures are inadequate for this usage. The model is limited in application to field situations that can be reasonably approximately by two-dimensional flow. The model applications described in Appendix G, however, provide insight into the difficulties associated with attempts at quantifying flow and contaminant

migration in fractured media of the type that occur at the Upper Ottawa Street landfill site. Except for most of the modelling study, the funds for the investigations described in this report were provided to the University of Waterloo by the Ontario Ministries of Health and Environment by way of the Upper Ottawa Street Landfill Site Committee.

To our knowledge the groundwater study described in this report represents the most comprehensive investigation undertaken to date at any Canadian landfill site on fractured rock. Some of the methods, conclusions and problems described herein have relevance to other landfills situated on bedrock in Ontario and elsewhere.

SUMMARY

The Upper Ottawa Street landfill, which occupies 18 hectares, is situated on the hard, cherty dolomite of the Lockport Formation. In the vicinity of the landfill this dolomite unit ranges in thickness from about 5 to 10 m. Shale of the Rochester Formation, which is about 7 to 10 m thick, exists beneath the dolomite. Below the shale, there are various units of limestone, shale and siltstone and then a thick sequence of shale strata of the Queenston Formation. All of the bedrock strata mentioned above have fractures, most of which are nearly horizontal and some of which are closer to vertical. The permeability of the strata can be attributed nearly exclusively to these fractures.

Data obtained in 1979 from monitoring wells in the landfill, which were reported in a previous study, and data obtained during our investigation indicate that a water-table mound has existed in the refuse at least since 1979. There has been no apparent decline in the level of the mound since 1979. The elevation of the top of the mound in the refuse is higher than the elevation of the water table in the bedrock at locations around the periphery of the landfill. Leachate from the landfill is therefore seeping into the bedrock around the landfill periphery. Beyond the periphery of the landfill, the regional slope of the water

table is generally eastward towards Lake Ontario. Therefore, although leachate can move outward from the landfill in all directions, it cannot travel far in any direction other than eastward.

Continuously rock-cored boreholes, ranging in depth between 18 and 58 m, were drilled at 26 locations outside of the landfill. Multilevel monitoring devices that consist of an assembly of narrow-diameter piezometers, ranging between 4 and 7 in number, were installed in 25 of these boreholes. Each piezometer monitors a different depth interval in each borehole. Each depth interval is isolated from the intervals above and below by inflated chemical packers covered by a rubber membrane. The position of each interval was selected on the basis of the apparent occurrence of open fractures indicated by the rock cores.

Water-level monitoring and water-level response tests conducted in the piezometers and in an open borehole and pumping tests conducted using the open borehole with monitoring in nearby piezometers indicate that the cherty dolomite that comprises the shallow part of the bedrock is moderately permeable (10^{-2} to 10^{-3} cm/s). The fractures in this zone are extensive and are generally well connected.

The Rochester shale beneath this zone is much less permeable. Hydraulic conductivity values from the piezometer response tests are typically in the range of 10^{-6} to 10^{-9} cm/s. A pumping test conducted in this shale zone near

the landfill, however, showed a major local horizontal hydraulic connection in the fractures in this zone and appreciable vertical hydraulic connection.

The limestones siltstone and shale strata beneath the Rochester shale exhibited a wide range of permeability. The limestone is generally more permeable than the overlying Rochester shale.

Vertical profiles of hydraulic head in the multilevel piezometers generally showed much higher levels in the shallow cherty dolomite above the Rochester shale than in the limestone or in the other units below this shale. This head differential, along with the much lower values of hydraulic conductivity from piezometer response tests in this shale indicate that the shale is generally quite resistive to downward groundwater flow. It is expected therefore that most or nearly all of the leachate from the landfill initially migrates laterally away from the landfill in the shallow flow regime in the dolomite, which varies in thickness from about 5 to 10 m.

It is known, however, that some vertical or near vertical fractures occur in the Rochester shale and that vertical hydraulic connection exists locally at the pumping test site. It is reasonable to expect that vertical hydraulic connection also exists elsewhere in the vicinity of the landfill. It is concluded, therefore, that a fraction of the leachate emanating from the landfill moves downward

through the Rochester shale into deeper bedrock zones. Near the landfill this fraction is probably small but the magnitude cannot be estimated. The fraction apparently increases eastward of the landfill towards Albion Falls.

From the nature of the land-surface topography and its effect on the slope of the water table in the shallow permeable dolomite zone, it is expected that much of the leachate from the landfill migrates towards Albion Falls. Inspection of the cliffs and ravine at Albion Falls, however, indicates that there is very little discharge of groundwater from the dolomite or from the contact zone at the top of the Rochester shale. The minor seeps that are evident in this area are derived from areas very nearby. The bulk of the leachate-impacted groundwater from the landfill that flows laterally in the dolomite towards the Falls probably migrates downward through the Rochester shale in the vicinity of the Falls. Greater downward flow near the Falls is expected because the shale in this area is probably more intensely fractured due to local rock stress conditions. Leachate-impacted groundwater that makes its way downward through the shale into stratigraphically lower bedrock units may travel in the subsurface all the way to Lake Ontario or it may emerge at surface somewhere between the bottom of the Albion Falls ravine and Lake Ontario. ← Monitoring devices were not installed in this area between Albion Falls and Lake Ontario because it was not considered practical to con-

duct monitoring in sufficient detail to trace the contaminant migration zone in the large area.

Calculations of the long-term average rate of leachate emanation from the landfill suggest a rate of about 2.3 L/s (30 gallons per minute). This estimate is based on the assumption that the existing cover on the landfill allows a moderate amount of infiltration. This rate is large relative to what is removed by the leachate collection system at the landfill which was determined in 1982 to be collecting less than a few percent of this amount. The seeps of leachate along the sideslope of the landfill along Redhill Creek are very small. No direct leachate seepages have been observed in the bottom of Redhill Creek. No leachate is known to be discharging as seeps along the Albion Falls ravine. Therefore, most of the leachate generated in the landfill apparently moves through the bedrock towards Lake Ontario. As this movement occurs some degree of attenuation of contaminant concentrations must take place because of dilution caused by dispersion. Additional attenuation may also occur because of chemical and biochemical processes. The degree to which the groundwater zone in the bedrock has been found to be impacted by the landfill is described below.

Data acquired since 1979 from piezometers in the landfill show no indication of a decline in the water-table mound. This indicates either that the cover has not appre-

ciably reduced infiltration of rain and snowmelt into the landfill during the past few years or that subsurface leakage of leachate out of the landfill is so slow that there is a long lag time between infiltration reduction and mound dissipation. If the cover were to be constructed in such a fashion as to make it impermeable, the water table would eventually decline to a level near the bottom of the landfill. Lateral eastward groundwater flow through the landfill would, even under these conditions, however, continue to cause leachate to be carried from the landfill in the shallow permeable dolomite zone, but the rate of leachate emanation would be smaller.

Groundwater samples were collected and analysed to determine the impact that the landfill might have upon the chemical quality of groundwater at this site. After installation, piezometers were periodically flushed to remove drilling water. Immediately prior to sampling, at least one or two well volumes were removed (where permeability permitted sufficiently rapid recharge) to ensure that water representative of the formation was obtained.

A broad range of inorganic and organic chemical parameters was employed to maximize the possibility of observing adverse effects of the landfill leachate upon groundwater quality. Often slow piezometer recharge rates precluded collection of sufficient water for complete analysis and so a sequence of collection priorities was adopted. Sampling

and analytical protocols were established with sufficient quality control to ensure that meaningful chemical data were obtained.

Groundwater in the study area generally contains large quantities of dissolved organic matter, typically 5 to 1500 mg/L dissolved organic carbon (DOC). Only a fraction of this DOC can be characterized by gas chromatography/mass spectrometry (GC/MS) but this fraction includes most of the organics presently considered hazardous. The rest of the organic matter is believed to be dominantly large molecular weight, very complex humic substances. Even for the organics studied by GC/MS, a large number of compounds present in groundwaters were not identified due either to chromatographic limitations such as coelution of peaks or to a lack of information on the mass spectra of compounds. The practicality of this technique precluded analysis of more than about 30 samples and most organic compounds identified in these samples were not quantified.

Some compounds are volatile or semivolatile and are termed purgeable organics. Volatile aromatic hydrocarbons including benzene and toluene and common industrial chemicals, especially solvents such as 2-butanone (methyl ethyl ketone or MEK) and tetrahydrofuran (THF) were found. Some of the toluene, MEK, THF and other purgeable organics could be derived from the piezometer material. Halogenated hydrocarbons such as dichloromethane, 1,1,1-trichloroethane and 1,3-

and 1,4-dichlorobenzene were found but in rather low concentrations, usually less than 20 ug/l, even in landfill leachate.

Non-volatile organics (termed extractable organics) were separated into acid and basic/neutral fractions prior to analysis by GC/MS methods. The base/neutral extractable organics most commonly found in groundwater samples included naphthalene, a polycyclic aromatic hydrocarbon, nitrogen- and sulphur-containing heterocyclic compounds, especially benzothiazole, and phthalate esters which are common plasticizers. The acid fraction is dominated by carboxylic and aromatic acids, but phthalates and phenol are also common. At least eight of the compounds identified in some groundwaters and landfill leachate are recognized pollutants, namely naphthalene, phenanthrene, acenaphthene, fluoranthene, diethyl-phthalate, di-n-butyl phthalate, bis-(2-ethylhexyl) phthalate and phenol. In groundwater and even in landfill leachate these compounds are usually present only at trace (< 10 ug/l) levels. Also, many of these compounds could have been derived from piezometer materials, especially the benzothiazole, some acids and phthalates. Once again, there is a general lack of extractable halogenated organic compounds.

Landfill leachate and groundwaters were also analysed for several inorganic parameters, including major cations (Ca, Mg, Na, K) and anions (Cl, HCO_3 as alkalinity, SO_4 , F),

nitrogen species (NO_3 , NO_2 , reduced Kjeldahl N and NH_3), boron, metals (Fe, Mn, As, Cd, Cr, Cu, Ni, Pb, Se, Zn), dissolved oxygen and methane gases. Landfill leachate generally has high concentrations of most cations, anions (except for low SO_4 values) and boron, irregular concentrations of metals, considerable dissolved methane and little dissolved oxygen and nitrogen is almost exclusively in the reduced form. Unfortunately, many of these characteristics are shared by some natural groundwaters at this site. In fact, some natural groundwaters contain considerably greater concentrations of major ions than do landfill leachates.

Volatile aromatic hydrocarbons (benzene, toluene, xylenes, etc.) were found to be a significant and readily determined component of landfill leachate and were therefore analysed quantitatively in many groundwaters. As with the inorganic parameters, natural sources exist, and so many uncontaminated groundwaters also contain significant concentrations of these organics.

The spatial distribution of the inorganic parameters and aromatic volatile hydrocarbons was generally very irregular and clear trends related to landfill leachate impact could not be seen. This reflects the complex hydrogeology of the fractured bedrock, the geochemical variability of natural groundwater in this area, some contamination of samples from piezometer materials and for some chemical parameters, the probable variation in leachate concentrations.

In order to assess the presence and perhaps degree of landfill leachate contamination in groundwaters a number of chemical criteria were applied to the chemical analyses. These included:

1. K/Mg ratio
2. Cl/SO₄ ratio
3. Ca/Alkalinity ratio
4. presence and approximate concentration of volatile aromatic hydrocarbons including chlorobenzene
5. presence of chlorinated organic compounds
6. presence of the common solvents THF, MEK and acetone
7. presence of benzothiazole, polychlorinated biphenyls (PCB's), phthalates, ethers, carboxylic and aromatic acids.

This provided fourteen tests, some of which could be applied to most groundwaters sampled. Application of these criteria resulted in low "scores" for groundwaters considered unimpacted by landfill leachate and higher "scores" for landfill leachate and obviously-impacted groundwaters. Groundwaters were also considered in terms of their apparent age (time since recharge, evaluated from the tritium content of water), the possibility of leaching of piezometer material as the source of apparent contamination, and the hydrogeological setting. This resulted in 26 of about 95 groundwaters being identified as contaminated. Some may have a non-landfill source of contamination. Most contamination

appears to occur within 100 m of the landfill, and is more commonly encountered on its east side. Elsewhere, contamination often occurs at depth and to the south and east of the landfill site (see Figure 38, Part 2).

The impact of landfill leachate upon groundwater quality, judging by available standards and criteria appears to be minor. This reflects the poor quality groundwater already present in bedrock at this site, as well as the limited criteria available to evaluate organic aspects of groundwater quality. This minor impact must be considered within the uncertainty of the identify of many organic compounds in groundwaters at this site and the undefined environmental consequences for many of the organic compounds that were identified in leachate and groundwaters.

Groundwater contamination at the Upper Ottawa Street landfill site is occurring but does not seem to result in a significant migration of contaminants from the groundwater zone into surficial soil zones or into surface water zones in a manner that could have an adverse effect on the neighborhood. Recognized contaminants are entering the groundwaters from this site, but, for the few contaminants considered, this does not seem to pose a serious long-term threat, particularly in view of the already-poor quality of much of the groundwater at this site and in view of the fact that, as far as known, there are no users of well water in the area.

Although we do not perceive groundwater contamination in the vicinity of the landfill as representing a cause of degradation of the environment at the land surface, it should be recognized that the landfill contains a very large volume of leachate. From information of the water-table level in the landfill, the depth to the bottom of the refuse and the landfill area, it is estimated that approximately 1.3 billion litres of leachate exist below the water table in the landfill. Except for the small fraction of this volume that will be collected in the leachate collection system, this quantity of leachate will eventually enter the fractured bedrock and migrate in the rock towards Lake Ontario. This volume will be augmented by the leachate that will form in future years or decades from rain and snowmelt that moves through the cap on the landfill. At present there are not many groundwater monitoring devices in the landfill and therefore a detailed characterization of the spatial variability of the chemical composition of the leachate is not available. The temporal changes in leachate composition are not known. These issues are appropriate for pursuit in follow-up studies to the one described in this report.

To put 1.3 billion litres of leachate in perspective, the following illustrative calculation is used. For this calculation it is assumed that the zone of contamination emanating from the landfill in the shallow bedrock is 1000 m

wide and that it extends with this width all the way to Lake Ontario, a distance of about 9 km. It is assumed that the thickness of this zone of leachate contamination is on average 30m. The void space in which water flow in the rock occurs is assumed to have an average value of 10^{-3} m^3 per m^3 of rock, which is a reasonable value for dolomite and shale, based on results of this study. The total volume of void space in this mass of rock between the landfill and Lake Ontario is 270 million litres. Thus, if contaminants migrate only in the fractures and do not enter the matrix of the rock by molecular diffusion, only 270 million litres of leachate would need to enter the bedrock in order to cause a plume of contamination to stretch all the way to Lake Ontario. Although the values used in this calculation are only estimates, it is reasonable to expect on the basis of any reasonable selections of rock parameter values that the 1.3 billion litres estimated to exist in the landfill would be sufficient to produce a plume that would extend to Lake Ontario.

If the rate of leachate production due to infiltration through the landfill cover is about 2.3 L/s as indicated above, the annual rate of leachate generation would be 70 million litres. Thus, only a few years of leachate production may be sufficient to cause the development of an extensive plume in the fractured bedrock.

RECOMMENDATIONS

The volume of leachate in the landfill is very large and the annual rate of leachate production due to infiltration through the landfill cover may be considerable. The number of monitoring wells in the landfill is insufficient to adequately characterize the spatial and temporal variability of the chemical composition of the leachate. We recommend, therefore, that about 5 to 10 additional monitoring wells be installed in the landfill and that these wells, along with the existing wells in the landfill, be used for periodic sampling and water-level monitoring for a considerable number of years. Depending on the results of such monitoring during the next several years, it may be appropriate to extend the monitoring time for an indefinite period. It would be appropriate for about 5 wells to be installed initially and once chemical and water level data have been acquired from these wells, there will be a better basis for making decisions regarding the exact number and locations of additional wells. These wells should be constructed entirely of stainless steel so that they will be less susceptible to damage than plastic wells and so that they will be excellent for studies of trace organic compounds of all types. These wells in the landfill, in combination with the existing wells, will provide a means of establishing detailed

trends in rise or decline of the water table in the landfill in response to changes in the condition of the landfill cover and in response to wetter or drier years.

Long-term monitoring of groundwater quality should be undertaken at this site. The goals should be to further characterize the organic components and to continue to monitor the inorganics and organics emanating from the landfill via the groundwater pathway. Specifically four additional piezometers should be installed near four sites, UW7, UW9, UW12 and UW22. Contamination has been confirmed at these locations and the existing core logs permit locating of additional piezometer screens in fractured, conductive zones which are transmitting leachate. Piezometers should be multiple installations, one shallow into the upper dolomite (< 15 m depth) and one deep into the lower limestone (20 - 25 m) at each site. The piezometers should be stainless steel, 5 to 8 cm diameter with screens of 2 to 3 m, sand-packed and isolated by grout or bentonite. The screens should be located in horizontally-fractured zones defined from observation of existing core or core logs. Sampling should be done using a gas-operated squeeze pump or peristaltic pump to flush the piezometer and samples taken with a teflon or metal bailer. Initially, samples should be exhaustively characterized for organic contaminants and then the concentration of selected, potentially toxic or toxic organics and inorganics determined at least twice each year. This will

provide some assurance that the contaminants of interest or of concern are properly identified and their flux to the groundwater adequately monitored.

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PART 1

Physical Hydrogeology of The Upper Ottawa Street
Landfill Site

INTRODUCTION

The Upper Ottawa Street landfill is situated on the plateau land at the top of the Niagara Escarpment about 5 km from Lake Ontario (Figure 1). The Niagara Escarpment owes its origin to an uppermost bed of hard cherty dolomite, which is more resistant to erosion and weathering than the other strata within the sequence of rocks along the escarpment. The landfill is located on this cherty dolomite, which is known locally as the Ancaster chert bed, and which is part of the Lockport Formation.

Prior to development of the landfill, the bedrock at the site was apparently covered by a deposit of sandy-silty glacial till that was a metre or two thick. In parts of the landfilled area, the till was removed when quarries were excavated into the rock. The quarries were later filled with waste. In other parts of the landfilled area, the layer of sandy silty till apparently was removed for use as cover material for the landfill. Thus, in a considerable part of the landfilled area, the waste water was deposited directly on the cherty dolomite, which is permeable because of fractures.

From previous hydrogeological studies of the site conducted by Gartner Lee Associates (1979) for the Regional Municipality of Hamilton-Wentworth, it is apparent that

there is a water-table mound within the landfill. It was evident therefore that leachate from the landfill is moving into the fractured bedrock. The extent to which leachate has travelled in the bedrock depends primarily on the hydrogeologic characteristics of the bedrock.

The purpose of this part of the report is to describe the geology, hydraulic conductivity and distribution of hydraulic head in the fractured bedrock in the vicinity of the landfill and to derive from this information expectations regarding the directions and rates of groundwater flow away from the landfill in the fractured bedrock. These expectations are used in Part 2 of this report in the identification of contaminated groundwater zones in the bedrock and in the consideration of possibilities for the existence of contamination at locations where the direct chemical evidence is inconclusive.

HISTORY OF LANDFILLING AT THE SITE

To develop information on the manner in which use of the Upper Ottawa Street landfill evolved from its earliest period, all available stereoscopic aerial photographs were obtained from the Ontario Ministry of Natural Resources. Aerial photographs taken in 1954 and in 1972 show the region at a scale of approximately 1:15000. The most recent aerial photographs, taken in 1978, are at a scale of 1:10000. An interpretation of the sequence of use of the various segments of the landfill area based on the aerial photographs referred to above are shown in Figure 2.

In 1954, landfilling occurred only in a small area, designated as area 1 in Figure 2, within the western part of what is now the entire landfilled area. Landfilling in this small area may have begun a few years prior to 1954. Also shown on the 1954 aerial photographs is an old quarry situated to the north of the present landfill entrance. This quarry has since been filled with refuse.

Most of the expansion of the landfill took place between 1954 and 1972. Part of the expansion was accomplished by diverting Redhill Creek northward before filling the area that previously served as the flood plain for Redhill Creek. That landfilling was essentially completed by this time in the area designated as area 2 on Figure 2. At

this time, filling was in progress in area 3 adjacent to the railroad tracks along the east side of the landfill. The 1972 photographs also show that landfilling had begun at this time along the southern part of the site. From the 1972 photographs it appears likely that the surficial soils in this area were excavated prior to landfilling. The large area, designated on Figure 2 as being 'disturbed' in 1972, probably contained shallow deposits of solid non-organic wastes.

Not much lateral expansion of the landfill occurred between 1972 and 1978. Some new filling occurred along the eastern part of the site, area 5 in Figure 2, but most of the landfilling involved topping up of areas 3 and 4. The 1978 aerial photographs show some lagoons, which apparently were used for percolation and fixation of liquid industrial wastes. The two northwestern ponds contained liquids in 1978. The southwestern ponds did not contain liquids in 1978 but they appeared to have been recently used for the disposal of liquid waste. The other ponds that are evident on the 1978 photographs are indistinct in outline and appear to have been used much before 1978.

Landfilling operations at the site ceased in 1980 and the entire landfilled area was covered with layers of flyash and soil.

FIELD METHODS

Drilling and Coring

A tracked-vehicle mounted CME 75 drill was used for drilling and bedrock coring at 25 sites (Figure 3) in the vicinity of the Upper Ottawa Street Landfill. Each site was drilled by augering through the overlying clay till to bedrock using hollow stem (10.8 cm ID) augers. Casing was set approximately 0.6 m into the bedrock. Bedrock was then continuously cored using NVL core barrels of 1.5 m and 3.0 m lengths. At the 26th site a 45° angle hole was continuously cored through bedrock under the landfill site using a Longyear P38 drill. Using the auger rig two holes were also drilled through the landfill into the upper zone of rock at the base of the landfill.

Water was used during coring of the bedrock, both to cool the drill bit and to remove the drill cuttings from the borehole. The drilling water was taken from local fire hydrants, which received water from Lake Ontario. The water used during drilling was not recirculated in the hole as coring took place.

Core removed from each hole was described in the field with respect to geologic and hydrogeologic characteristics. Characteristics of geologic units and location of fracture zones were recorded. The orientation, size, infilling, and

staining of fractures were noted. From examination of the core, zones with apparent higher hydraulic conductivities (i.e. larger or more frequent fractures) were selected for placement of sampling intervals in multilevel monitoring devices.

Installation of Multilevel Devices

A detailed description of the multilevel sampling device is provided in Cherry et al. (1985) and in Appendix F of this report. Figure 4 illustrates the components of the device. The 25 boreholes that were equipped with the device were drilled in phases, each phase including several holes. Some of the details pertaining to the materials and design of the device changed from phase to phase as problems were recognized and improvements made.

From preliminary testing in November, 1981, it was felt that five monitoring points would be the optimum number of points for each multilevel device. Thus, five fracture zones were generally chosen for each core hole and a multilevel device was designed and assembled based on the location of these zones. Multilevel devices installed during the early phases of drilling had water sampling zones varying in length from 0.6 m to 1.2 m, depending on the size of fracture zones encountered. However, for later phases of drilling, water sampling zones were generally 1.2 m in length because many fracture zones, especially in the shale,

produced very small amounts of water, and it was decided that the longer sampling zone was necessary to obtain sufficient water.

Sample tubes and sample ports were assembled for each sample zone. Each sample port was located near the bottom of the sampling zone. This was done for two reasons; first, to facilitate attaching the sample point to the PVC casing and second, to enable removal of the entire column of water located in the sampling zone in order to obtain fresh formation water when sampling.

Upon completion of the installation, the inside of the PVC casing was filled with water. After the packers were given sufficient time to expand, water-level monitoring and sampling began.

Testing of the Multilevel Devices

The first multilevel device was installed in November, 1981 and was tested briefly to ensure that the packers were performing properly.

This first device was tested in two ways: first, by response testing each sampling zone and measuring water levels in adjacent sampling zones and second, by chemical analyses of water samples from each zone for specific conductance, total organic carbon and chloride.

On the basis of the testing performed, it was decided that the packers had performed successfully, that major

leaks could be identified, and further installation of multilevel devices continued at the Upper Ottawa Street Landfill in 1982.

Following installation of 19 multilevel devices at the Upper Ottawa Street Landfill during the spring, summer and fall of 1982, more detailed testing was done in the field to assess the performance of each device.

A solution of rhodamine dye was injected into the column of water in each of the PVC casings. The dye was given sufficient time to mix and the water in the PVC casing was tested to ensure mixing had taken place.

At most installations, water levels were monitored in the piezometer tubes and in the PVC casing before, during and after injection of the dye. Fluctuations of water levels in response to the injection would indicate hydraulic connection between the PVC casing water and the piezometer, thus signifying leakage was occurring. After the dye solution was injected, piezometer tubes were flushed using the sampling methods described in the next section. This flushing was done to determine if dye was present in any of the well points.

Observations of dye indicate that some leakage of water from inside the PVC casing (centre column) into some sampling zones was occurring in some sampling zones. Water level measurements taken in sampling tubes during dye injection showed water level changes in few sampling zones.

Pumping of sampling zones after dye injection in the centre column showed dye present in a small percentage of the piezometers.

Changes in water levels in sampling zones during injection would suggest the water level in the sampling zone was responding to an increase in water level in the centre column, indicating a hydraulic connection between the centre column and the sampling zone. In the multilevel devices tested, the zones which showed changes in water levels were generally the low hydraulic conductivity zones (i.e. slow responding zones). This is probably because the water storage area in the fractures is small and it would take very little leakage of water to change the water level. In contrast, high permeability zones would not necessarily demonstrate a change in water level because the effect of leakage could dissipate quickly.

The multilevel device was further tested in the laboratory to determine whether the points of leakage could be detected. Segments of PVC casing separated by packers were placed in 7.5 cm (3 in) ID plexiglass tubes and the centre column was filled with water to determine the quantity and location of leaks. It was found that when leakage was occurring in the sampling zone it was generally at the point where the elbow on the piezometer port protruded through the casing. Some packer leakage occurred during the initial expansion of the Dowell seal sleeve. The seals became water tight after a few days.

Both of these problems have since been rectified by changes in design of the piezometers that were installed later in the drilling program. These changes are described in detail in Appendix G.

The final stage in the preparation of the multilevel devices for stable, long-term monitoring involved filling the centre column of each of the devices with a bentonite slurry. The purpose of the bentonite slurry is to seal any leakage points that may exist at the couplings, at the piezometer elbows or along the inside of the packers. This stage, in which the first 16 multilevel devices were filled with bentonite slurry, was conducted in November, after all of the multilevel devices that were planned for 1982 had been installed. The remaining multilevel devices installed in 1983 were filled with a bentonite slurry after preliminary monitoring of each was completed.

Hydraulic Conductivity and Storativity Determinations

Hydraulic conductivities were determined by three methods, response testing of individual sample zones or piezometers, borehole packer testing discrete intervals within an open borehole, and pumping various zones within an open borehole and monitoring water level responses in wells surrounding the pumping well. A detailed discussion of the methods and results is given in Appendix G.

Response testing individual piezometers for hydraulic conductivity was performed using a method developed by Hvorslev (1951). This involves removal of water from (rising head test) or addition of water to (falling head test) a piezometer. The rate of water level rise or decline back to equilibrium water level is monitored for each piezometer, and in this way the local horizontal hydraulic conductivity can be determined in the vicinity of each piezometer.

Hydraulic conductivities were determined at UW25, the location of which is shown in Figure 3, by pressure packer testing discrete intervals within the open borehole. Packer testing involves isolating an interval within a borehole by a device known as a packer. The double packers used for testing in this study were pneumatically inflated with nitrogen, to expand against the borehole wall and isolate the zone between them. Water was then injected into this zone at a constant pressure and flow rate.

A series of pumping tests were performed at UW25 and water level responses were measured at numerous multilevel devices in the vicinity of UW25 in order to determine the hydraulic conductivity and storativity of the units being tested as well as examine the degree of fracture connection both vertically and horizontally within these units.

Pumping tests were conducted using a gasoline-powered suction lift pump to remove water from the borehole. Discrete zones were tested by isolating the zone with pneumatic

packers as previously described, and pumping waters from this zone. Water level responses were monitored at locations in the vicinity of the pumping wells (Figure 61a). Standard methods of analyses were employed to determine hydraulic conductivities and storativity and the methods and results are discussed in Appendix G.

RESULTS AND DISCUSSION

Hydrostratigraphy

A simplified vertical geologic section is shown in Figure 6 and Figure 7 and 8 show geologic cross-sections in the vicinity of the landfill. Locations of the cross-sections are shown in Figure 3. Figures 9, 10 and 11 show representative borehole logs of the study area.

The uppermost geologic unit consists of glacial drift. As encountered in the University of Waterloo borings, this drift comprises glacial till and glaciolacustrine clay. It is generally less than 3 m thick except in the moraine located north of the landfill, where the thickness is about 12 m.

Below the glacial drift, the uppermost bedrock unit encountered was a dolomite to a shaly dolomite found west of the small scarp on which the landfill is located (i.e. UW5, UW15, UW11). This unit is generally described two ways with the uppermost part of the unit generally a dolomite with shaly sections and the lower part of the unit a shaly dolomite with some zones being termed as shale. The unit corresponds to the Eramosa member of the Lockport Formation as described by Bolton, 1957. The unit is described as being a dense thin bedded dolomite that is petroliferous, with bituminous shale partings. The upper zone is more massive,

which would correspond to the dolomite with shale zones (i.e. UW5). The unit, where present, is up to 20 m in thickness.

Although the upper unit is more massive, some intervals indicate large fractures present, probably along the bedding planes. This was indicated during coring of UW5, by water cascading down the drill hole from shallow zones. Most fractures, however, were tight, usually occurring along shale partings. Fractures encountered were horizontal or nearly horizontal. Rock core examined from this unit indicates that the rock itself (i.e. the rock matrix) will have a low hydraulic conductivity but several fracture zones indicating a high hydraulic conductivity were found in each borehole.

The rock unit immediately below this unit west of the landfill is a cherty dolomite. This cherty dolomite is the uppermost unit east of the landfill (Figure 5). This cherty dolomite is the Ancaster chert beds which form the base of the Goat Island Member of the Lockport dolomite. The unit was encountered at every location drilled. This is a massive, irregularly bedded fine grained dolomite with a high argillaceous content. Many vugs are present and are often filled with calcite and gypsum. This unit forms the cap rock for much of the Niagara Escarpment in the Hamilton area. The distinctive nature of this cherty dolomite makes it a useful marker throughout the area. It is referred to as cherty dolomite on all borehole logs.

The cherty dolomite encountered during coring at each location indicates that the unit is massive but is well fractured along the bedding planes. Fractures were generally horizontal or near-horizontal and occurred more frequently as fracture zones, rather than isolated fractures. Some vugs were also present and were filled with calcite and gypsum. One or two larger fracture zones were usually encountered at each site. These were the zones of highest hydraulic conductivity encountered during drilling, as evidenced by large water losses of the circulating drilling water.

Underlying the cherty dolomite is what has been described as a shaly dolomite and dolomite in borehole logs. Although shown on the drill logs as a single unit, this may in fact be two units as described by Bolton (1957). These are the Gasport Member of the Lockport Formation and the underlying Decew Formation. At the Upper Ottawa Street Landfill this unit has a highly variable lithology ranging from a semi-crystalline dolomite to finely-bedded shaly sections up to 1.0 metre thick. Alternating thinly bedded semi-crystalline and shaly zones were often encountered during drilling.

Very few fractures were encountered in this shaly dolomite unit. Most fractures were horizontal, occurring along the shale bedding planes and were very tight, indicating a relatively low hydraulic conductivity throughout this

unit. The only significant fractures were found in the less shaly upper sections of this unit which is probably the Gasport Member.

Below the shaly dolomite is a shale unit known as the Rochester shale which is generally 7 - 10 m thick. It consists of dark calcareous shale and siltstone with some thin limestone beds. The upper zone is generally a massive dolomitic or calcareous shale. The shale is slightly or moderately fractured, generally along the horizontal bedding planes. Many of the fractures are infilled by clay and occasionally calcite and gypsum. Some shale zones encountered were up to 2.0 metres thick without any noticeable fractures. Limestone beds up to 0.3 m thick were encountered in the Rochester shale. These units contained some fractures which were generally horizontal, very tight, and did not appear to have any zones of high hydraulic conductivity.

Below the Rochester shale is a thin limestone unit, generally only a metre thick, known as the Irondequoit Formation. It is a crystalline porous limestone.

The unit underlying the Irondequoit is known as the Reynales Formation. It is described in the borehole logs (Figures 9 to 11) as limestone, shaly limestone or porous dolomite, and is generally 2-4 metres thick. It is described by Bolton (1957) as a dense dolomite with grey shale partings with a basal zone of dense limestone. This

unit had several horizontal fractures present at numerous drilling locations.

The unit underlying the Reynales Formation is described in different ways as seen in the borehole logs. It is described as a shaly dolomite, shale, siltstone and shale and ranges from 5 to 10 metres in thickness. This unit is probably two units referred to by Bolton (1957) as the Thorold Formation and the Grimsby Formation. The Thorold Formation is described as a sandstone, with numerous thin green to grey shale and siltstone partings. The Grimsby Formation, also called a sandstone, is locally described as a red, fine grained shale to siltstone with red interbedded shale increasing downward.

These units have numerous shale partings and although they appear tight in drill core, they possibly have a higher hydraulic conductivity than most shaly zones encountered.

Below these units, encountered at a few sites drilled deeper, is a unit of alternating shale and limestone beds with some sandstone (Figure 8) known as the Cabot head Formation. Rock core examined from this unit indicates a relatively high degree of fracturing in the limestone sections.

Hydraulic Conductivity

A detailed discussion of the results of hydraulic-conductivity testing of the bedrock is presented in Appendix G. Only a brief assessment is provided here.

The water-level response tests in the piezometers gave a wide range of hydraulic conductivity values (table 1) and produced various shapes the water-level recovery vs. time relations. The response tests provided hydraulic conductivity values that were generally greater than 10^{-4} cm/s in the cherty dolomite within 10m of ground surface. The zone is therefore moderately permeable, in the range equivalent to a fine to medium-grained sand aquifer. The tests in this zone gave log water-level recovery vs. time relations that were typically linear. The magnitude of the hydraulic-conductivity values and the linearity of the response indicate that this zone has numerous open fractures. Because of the linearity of the response the data were well suited for calculation of hydraulic conductivity values using the method of Hvorslev (1981).

For the Rochester shale, the piezometer response tests gave hydraulic conductivity values that were generally low, between 5×10^{-6} to 1×10^{-9} cm/s. This suggests that this zone is only slightly permeable. This range is equivalent to the hydraulic conductivity of unfractured silt and clay. The log water-level recovery vs. time relations were of two types: linear and non-linear. The linearity occurred primarily during the early period of recovery. The piezometers with the linear response generally provided lowest hydraulic conductivity values, less than 10^{-8} cm/s. The shale at these piezometers apparently has very small but extensive

fractures. These fractures are probably along horizontal bedding planes. The piezometers that had non-linear responses probably intersect two types of fractures, the small extensive ones and larger very local ones.

A small number of piezometers in the Rochester shale provided values of hydraulic conductivity that were much larger than the typical range described above. The highest value was 1.5×10^{-3} cm/s, which suggests that there are local zones in the shale that are much more fractured than is the norm for the bulk mass of the shale. This conceptualization was confirmed by pumping tests described below.

The water-level response tests conducted in the open borehole at UW21 using gas-inflatable packers to isolate the test intervals in the hole gave similar results for the cherty dolomite to those described above for the piezometer tests. At this site, the Rochester shale is very permeable ($K > 1 \times 10^{-3}$ cm/s).

The values of hydraulic conductivity obtained from the piezometer response tests vs. depth are shown in Figure 12. This graph indicates that although there are numerous high values at shallow depth and many lower values at greater depth, some high values also occur at depth. Many of the higher values at depth occur in limestone or siltstone beds that are beneath the Rochester shale.

Results of the pumping tests, all of which were conducted from the vicinity of UW21, indicate that although the

ground water flow system is complex and parts of the system respond in a very non-homogeneous manner, the hydraulic conductivity determinations are quite consistent within the zones that responded to pumping. Because much of the monitoring network consisted of multilevel installations, water level responses could be examined in various discrete vertical zones. Results for the main water bearing zone, the Lockport dolomite, were obtained from fifteen discrete zones within the unit. Fifty-one values of hydraulic conductivity were obtained from four pumping tests using seven different methods of data analyses. Hydraulic conductivity values ranged from 2.6×10^{-2} cm/sec to 1.1×10^{-3} cm/sec, showing a relatively consistent value of hydraulic conductivity within the Lockport dolomite. Zones which responded during pumping were generally the zones show high hydraulic conductivities from the results of the response testing. These values are also consistent with the values obtained from the piezometer response task. This indicates that the hydraulic conductivities of the rock at the scale of a few cubic metres or less near a piezometer interval is that of the bulk rock mass at a scale of many tens of cubic metres.

The zone showing one of the best response as a result of pumping was the high hydraulic conductivity zone in the low hydraulic conductivity Rochester shale found at separate locations during the response testing and packer testing. Values for response testing, packer testing and pump testing

(average) of this zone were 2.6×10^{-3} cm/sec, 1×10^{-3} cm/sec and 2×10^{-3} cm/sec, respectively.

The upper 10 m of rock downgradient of the landfill, which consists primarily of the cherty dolomite, exhibits the highest overall hydraulic conductivity of all the rock units. However, some of the underlying low hydraulic conductivity units exhibit zones of high hydraulic conductivity and the potential for significant water movement through these units relative to the upper rock units will depend on how well connected these zones are to each other.

Water-Table Configuration

Detailed monitoring of water levels in the University of Waterloo multilevel installations were performed during the present study to gain a better understanding of groundwater flow directions in the vicinity of the landfill. Water levels were monitored on a routine basis during the study, however, at some sampling locations in low hydraulic conductivity zones, water levels took up to two months to achieve equilibrium and measuring of this equilibrium was seldom achieved.

Most piezometers, however, were able to remain undisturbed for a sufficiently long interval to permit a number of equilibrium measurements as well as measure the winter/spring recharge events and the spring/summer decline.

Figure 13 shows a generalized water table contour map. The water table map is really a composite of water levels taken during the study and although the water levels may vary somewhat during the year (i.e. higher during spring recharge), the general trend remains the same. The water table in Figure 13 is an approximation of summer water levels after the decline of spring recharge.

Few water level data are available within the landfill itself. Water levels were also monitored within the landfill to examine water table mounding. Two previous boreholes were placed within the landfill by Gartner Lee Associates and two more boreholes were placed in the vicinity of the previous ones during the present study. Water levels within these boreholes fluctuated by several metres with a rapid rise in water levels during the late winter and early spring. Water level data from wells within the landfill indicate that the water table within the landfill has mounded significantly.

Mounding beneath landfills is a common feature in Ontario. This mounding is caused by greater infiltration and less evapotranspiration in the landfill area relative to areas away from the landfill. A cover has been placed on this landfill to reduce infiltration. The effectiveness of this cover as a means of limiting infiltration is not known. It will probably take several years to see a noticeable dissipation of the mound. Water level monitoring conducted

during the present study indicates that the mound has not begun to dissipate yet, three years after the clay cover was placed at the landfill.

Water levels were measured routinely in monitoring wells installed within the landfill. Water levels fluctuated by about 1.2 m at UW26, probably in response to recharge events, during monitoring from June 1983 to October 1984. There was no overall decrease in water level during this time. At UW28, water levels were more difficult to obtain due to gas generation (i.e. water was bubbling and frothy) however water level values are similar to those obtained by Gartner Lee between October 1977 and May 1981, which would indicate no dissipation in the water table mound to date.

With the exception of the groundwater mound, the water table map demonstrates that the lateral component of shallow groundwater flow is generally from west to east, with flow primarily directed towards Albion Falls Ravine to the east. Along the high clay till ridge to the east of the landfill there appears to be a small water table divide in the shallow groundwater zone. Some of this shallow water may discharge into Redhill Creek. The quantity of discharge is not known, however no seeps of contaminated water from the landfill were evidenced along the creek bottom. Some seeps or leachate breakouts along the side of the landfill were evidenced, however there was no noticeable difference in water quality upstream and downstream of the landfill (Gartner Lee

Associates, 1980). Work done by Gartner Lee Associates did not indicate any noticeable gain in stream discharge along this portion of Redhill Creek.

Vertical Hydraulic Head Distribution.

Hydrogeologic sections shown on Figures 14 and 37 show vertical profiles of water level data along cross sections A-A' and B-B', respectively. The vertical profiles indicate a number of trends with respect to groundwater flow in the vicinity of the landfill.

In the shallow groundwater zone (i.e. the cherty dolomite and underlying dolomite of the Goat Island and Gasport Members) adjacent to the landfill the vertical gradients are very small indicating primarily a horizontal movement of water. Moving downgradient of the landfill site (i.e. UW1, UW3, UW2) the vertical gradient increases within the cherty dolomite (Goat Island) with distance from the landfill. At the base of the cherty dolomite, in the massive dolomite of the Gasport and Decew units, little vertical gradient exists indicating primarily horizontal flow in this zone.

Upgradient of the landfill site (i.e. UW5, 10, 11, 15, 17, 18, 20) the shallow groundwater zone, (the upper 15 m) shows two distinct patterns. UW5, 10, 11, and 20 located a relatively great distance from the landfill, compared to UW15, 17, 18, show small overall vertical gradients, except in isolated zones primarily at the contacts of two rock units. Water levels found at these contacts, are often low-

er than in the units above, indicating relatively permeable zones at the contacts.

The second pattern shows that near the landfill strong downward gradients are found in the units above the cherty dolomite. This could be caused by the effect of mounding within the landfill or by the presence of the small scarp buried by the landfill acting as a drain for the upper rock units in the vicinity of UW17.

In the deeper groundwater zone below the cherty dolomite, the shale or shaly dolomite of the Rochester Formation shows a relatively strong downward gradient, in some cases almost unity (i.e. 1 m decline in water level for 1 m drop in elevation). The only exception to this is found immediately downgradient of the landfill site (UW1, UW3) in which there is an apparent upward gradient. These zones, however, were zones in which natural gas was encountered during drilling and it is possible that this may create an excess pressure in these zones causing higher water levels.

Underlying the Rochester shale, the limestone or porous dolomite of the Irondequoit and Reynales Formation appears to act as a drain at least in some locations, (eg. UW11 and 20). The water levels found within the unit is often considerably lower than above it. This would indicate that there is a significant component of horizontal groundwater movement, at least in some locations, in this rock unit. The direction of movement would be the same as the general

groundwater flow direction, west to east. The amount of water able to move through the Rochester shale into this unit will depend on the vertical connection through the Rochester shale.

In the vicinity of Albion Falls, water levels indicate that a "perched" water table exists within the cherty dolomite near ground surface. Water seeps along the rock face of Albion Falls show some seeps near the cherty dolomite and Rochester shale contact but it is generally dry almost to the bottom of the rock face. The water at the cherty dolomite Rochester shale interface is likely from water in the small pond above Albion Falls.

Water levels at UW19 indicate perched water in the shallow zone and below this there appears to be only a small upward vertical gradient at the base of Albion Falls indicating a small component of flow into the base of Albion Falls. Below the base, there is a stronger vertical gradient downward within what is described as shale with some dolomite, which probably corresponds to the Cabot Head Formation.

Recharge.

Water levels were monitored during winter/spring recharge and spring/summer decline to examine the extent of groundwater recharge with depth within the fractured rock in order to better determine the potential for water from the landfill site to migrate vertically downwards. During the

winter and spring of 1983, water levels were monitored for increases in response to recharge from snow melt and precipitation events. Because of the relatively mild winter in 1983, significant recharge occurred during the winter months as well as in the spring. Also, because of the very hot and dry summer of 1983, significant declines in water levels occurred in many areas due to lack of recharge, thus making water level trends more pronounced than might normally occur.

Table 2 shows water level fluctuations and approximate periods of fluctuation for wells which had adequate data to determine these fluctuations. A number of trends are apparent and will be discussed by examining different areas around the landfill site.

Areas immediately adjacent to the downgradient side of the landfill (i.e. UW6, UW7), and areas downgradient (i.e. UW1, 2 and 3) indicate similar water level rises and declines within the cherty dolomite and underlying dolomite (Goat Island and Gasport Formations). The general water level rise was about $40 \text{ cm} \pm 15 \text{ cm}$ and the decline was about $65 \text{ cm} \pm 15 \text{ cm}$. There was no apparent trend in increase or decrease in water levels with depth or variations in times of increase and decrease with depth indicating these units to be reasonably well connected. Water levels generally increased from February to April and declined in May through July. Variations in water level fluctuations within these

rock units may reflect local variations in recharge or variations in porosity of the rock.

Upgradient of the landfill (i.e. UW5, UW15) there appeared to be a more substantial recharge in the shallow groundwater zone. The water level rise and decline was generally greater in the upper 10 m of the groundwater zone, sometimes fluctuating more than 1.5 m. The water level fluctuation also indicated a greater depth of recharge upgradient of the site, on the order of 25 m, to the base of the cherty dolomite in some cases.

Nearer the landfill on the upgradient side, (i.e. UW17) water level fluctuations were significant, up to 4.0 m in some cases. This could be due to the increased infiltration associated with the landfill, as evidenced by mounding of the water table within the landfill. However, this large fluctuation in water level only occurred very near the landfill and would not have a significant effect on the overall groundwater flow patterns short distances (i.e. 100 m) from the landfill.

Below the cherty dolomite of the Goat Island Formation, the Rochester shale appears to show little evidence of recharge events affecting water level fluctuations. Although water level fluctuations occur, there appears to be no noticeable trend. However, of more importance and which may indicate potentially significant recharge through the Rochester shale are the large water level fluctuations in

some areas within the dolomite and limestone of the Irondequoit and Reynales Formations immediately below the Rochester shale. Fluctuations of several metres (i.e. UW6-5, UW13-3, UW13-4, UW14-2, UW14-3) indicate that recharge may be occurring into these units. The volume of recharge cannot be well defined because these lower units will likely be more confined than the upper cherty dolomite, and as a result it would take less recharge to cause a greater fluctuation in water levels than in the upper units.

The water level fluctuations indicate that although there does not appear to be significant recharge into the Rochester shale from above, the unit below the shale does appear to behave at some locations as if recharge were occurring through the shale, and the vertical gradients exhibited in the shale also show that potential does exist for this recharge to occur.

Vertical Interconnection Testing

Vertical interconnection testing was completed in August, 1983 on multilevel devices UW1, UW2, UW3, and UW21. The results of the testing are compiled for each multilevel device in table 2. The hydraulic conductivity, as obtained by the response tests, and the static water level of each piezometer in the multilevel device are noted in the table because this information aids in interpreting the results.

Usually several different piezometers at any one multi-level device were pumped out, and the results from each pumpout is listed on a separate line. The piezometer from which water was removed is marked by a TT, which indicates water was removed by a triple tube sampler, or by a PP, which indicates that the water was removed by a peristaltic pump. The volume of water removed from each piezometer is also listed. The drawdowns of the other piezometers at that multilevel device are expressed in meters. If the water level of a piezometer was not measured this is indicated by a NI (no information). If the water level of a point had not fully recovered from a previous pumpout, this is noted by an I (interference).

In several of the pumpouts, the water levels of adjacent piezometers dropped, signalling the presence of a vertical connection. These cases are underlined in Table 2. A vertical connection was noted between UW1-3 and UW1-4 because the level for UW1-4 dropped 9 cm when 975 ml of water were removed from UW1-3. This vertical connection could be a vertical leakage caused by a packer that has not fully inflated in the borehole. This is unlikely, however, because the static water level of UW1-4 is 12 cm higher than that of UW1-3. The vertical connection between UW1-3 and UW1-4 is probably the result of fractures.

UW2-3 dropped 6 cm when 50 liters of water were removed from UW2-2 by a peristaltic pump. The static water level of

UW2-3 is 8 cm lower than that of UW2-2. This difference is not great enough to totally discount packer leakage as the cause of the apparent vertical connection between UW2-2 and UW2-3.

Vertical connections were also shown to exist between UW3-2 and UW3-4, and between UW3-2 and UW3-3. UW3-3 and UW3-4 exhibited no evidence of being connected. As the vertical connection between UW3-2 and UW3-4 bypasses UW3-3 which is located between them, it is unlikely that this vertical connection is the result of packer leakage. This connection must be caused by a fracture system which is not vertically planar but rather is an irregularly connected network with the individual fractures being limited in extent.

No vertical connections were observed at UW21 or at the other piezometers tested at UW1, UW2, and UW3. This fact alone does not preclude the possibility of there being vertical connection because in zones where the hydraulic conductivity above or below the pumped interval is high, the pumping effect may not be large enough to be detectable. Or if the pumped interval has a very high hydraulic conductivity, the drawdown in this interval may not be large enough to cause a detectable drawdown in the other intervals. However, if both the point being pumped and the point being measured have a low or moderate hydraulic conductivity, the presence of any vertical connections should be apparent from the drawdown of the point measured.

Using this reasoning, the results of vertical interconnection testing are inconclusive for possible connections between: UW1-2 and UW1-4, UW2II and UW2-2, UW3II and UW3-2, UW3-4 and UW3-5, UW21-1 and UW21-2, UW21-2 and UW21-3, and UW21-3 and UW21-4. Also using this reasoning, no vertical connections exist between UW1-2 and UW1-3, UW1-4 and UW1-5, UW1-4 and UW1-6, UW3-3 and UW3-4, UW3-3 and UW3-4, UW21-3 and UW21-5, and UW21-4 and UW21-5.

The interpretation of vertical interconnection testing is clearly complicated by the many variables that need to be considered. One can say that two sampling points are connected if a change in water level is noted, but one cannot easily conclude that no vertical connection exists if no change occurs, without dealing with these complicating factors.

SUMMARY OF THE GROUNDWATER FLOW SYSTEM

Based on the previous discussion of results a general summary of the groundwater flow system is given here. Water table conditions indicate the general direction of groundwater movement is west to east towards Albion Falls. A water table mound has developed within the landfill. Although the mound is higher than the water levels upgradient of the site, the mound dissipates quickly near the boundaries of the landfill resulting in only a small component of groundwater flow from the landfill moving a short distance (i.e. 100 m) upgradient of the landfill. Although it is possible that some of the landfill derived water may move along the north-south trending scarp on which the landfill is located evidence to date indicates this is not very likely.

The bulk of the groundwater coming in contact with landfill derived leachate appears to move horizontally through the high hydraulic conductivity cherty dolomite (Goat Island Formation) on which the landfill is situated. Some very shallow groundwaters (i.e. upper few meters) and seepage breakouts along the eastern edge of the landfill discharge into Redhill Creek, adjacent to the landfill.

As the groundwater migrating horizontally through the cherty dolomite moves towards Albion Falls the vertical component of groundwater flow increases in the dolomite and

through the underlying Rochester shale. Although the Rochester shale and underlying rock units have a low hydraulic conductivity compared to the cherty dolomite above there is some degree of horizontal and vertical fracturing. The downward hydraulic gradients through the shale, coupled with isolated vertical fractures appear to indicate that the bulk of potentially contaminated groundwater can migrate downward prior to reaching Albion Falls. This appears to be substantiated by the lack of seeps or water discharge zones along the rock face of Albion Falls. Downward gradients below the base of Albion Falls at a multilevel monitor placed there also indicates that groundwater is likely moving deeper than the base of Albion Falls. This water will eventually discharge into Lake Ontario, approximately 5 km from the landfill, either directly or some of it may emerge in the bottom of Redhill Creek or along its banks farther downstream. It is unlikely however, that if leachate impacted water emerges along Redhill Creek beyond Albion Falls, it will be detectable or constitute a water quality problem.

The occurrence of leachate-impacted groundwater very near the landfill is difficult to detect because the natural groundwater has very poor quality and because the contaminants are not very diagnostic. At considerable distance from the landfill (i.e. further downstream along Redhill Creek) detection will be even more difficult because of dilution effects caused by dispersion and rock matrix diffusion.

Leachate Generation

The flux of leachate from the landfill can be estimated by two methods. The first method, which was used by Gartner Lee Associates (1979), is based on water budget calculations pertaining to infiltration on the landfill surface. It is assumed that the amount of leachate that leaves the landfill by seepage is equal to the water entering the landfill by infiltration. The second method consists of estimation of lateral groundwater flow for the shallow permeable zone in the cherty dolomite and the shaly dolomite zone at the periphery of the landfill. It is assumed that the leachate that enters the lateral flow zone in the upper part of the bedrock near the bottom of the landfill represents essentially all of the leachate that emanates from the landfill.

The water budget method used by Gartner Lee Associates is based on a comparison of average precipitation and evapotranspiration. From a 20 year record of climatological conditions at Mount Hope Airport, the average annual precipitation was estimated to be 80 cm (31.3 inches) and the average annual evapotranspiration was estimated to be 62 cm. For a vegetated soil, the difference between the precipitation and the evapotranspiration, which represents the average annual potential water surplus, would be 18 cm (7.1 inches).

Because the cover on the landfill is not vegetated, Gartner Lee Associates believed the water surplus would be larger than this calculated value. They selected a value of 28 cm (11 inches) and assumed that this amount of water, which is about 35% of the annual precipitation, infiltrates into the landfill over the entire landfill surface. This is equivalent to 12.7 L/min per hectare (0.5 gpm per acre). For the entire landfill, this amounts to 2.3 L/s (30 gpm).

In the second method the lateral flow of shallow groundwater away from the landfill is estimated using the Darcy equation,

$$q = KiA$$

where q is the groundwater flux per unit cross sectional area, K is the hydraulic conductivity and i is the hydraulic gradient. The flux calculation is made for a cross sectional area (vertical plane), which is 7 m deep (i.e. the approximate thickness of the cherty dolomite) and 600 m long. This distance represents the length of the northern and eastern boundary of the landfill along Redhill Creek. The lateral hydraulic gradient along the boundary is approximately 0.05. The values of hydraulic conductivity of the shallow bedrock zone (i.e. in the upper 9 m) are generally in the range of 1×10^{-3} to 1×10^{-2} cm/s. With these values the Darcy equation yields a total lateral flux in the range of 2.1 to 21 L/s (28-280 gpm). The total flux estimated by Gartner Lee Associates using the water budget method (2.3 L/s) is within this range.

The flux estimates obtained from both methods have considerable uncertainty. In the water budget method the main uncertainty pertains to the percentage of surplus water that leaves the landfill area by surface runoff. No measurements of surface runoff have been made at the Upper Ottawa Street landfill site or at any other landfill site with a similar surface. Surface runoff estimates for the Upper Ottawa Street landfill are even more problematic than would normally be the case because the landfill surface has gradually changed during the past few years as cover material has been applied and as contouring and small-scale erosion channels have developed.

There are several causes of uncertainty in the results of the groundwater flux estimates obtained from the Darcy equation. The groundwater flux is not actually an estimate of leachate emanating directly from the landfill but leachate contaminated groundwater. Groundwater from the landfill may flow deeper than the 9 m depth of the cross section. There are probably local zones on the cross section that have a higher or lower hydraulic conductivity than the range used in the calculation.

The water budget estimate and the Darcy-equation estimate are not directly comparable. The water budget estimate calculates leachate generated in the landfill based on estimates of water infiltration through the landfill cap to the water table in the landfill. The Darcy equation estimates

calculates the volume of groundwater that flows laterally through the cherty dolomite bedrock directly under the landfill. This water comes in contact with and mixes with leachate from the landfill and the Darcy estimate is actually leachate contaminated groundwater moving laterally away from the landfill. Because the low end of the range of volume of leachate-contaminated, water flowing laterally away from the site agrees well with the estimated volume of leachate generated at the landfill due to infiltration it is reasonable to expect that the volume of leachate generated is on the order of a few litres per second (25 gpm).

Even if the value of on-going leachate generation in the landfill is not considered, the volume of leachate present with the landfill is still substantial. The volume of leachate within the landfill can be calculated by considering, the size of the landfill, the portion of the landfill below the water table and the porosity of the landfill material.

The landfill is approximately 600 m long and 300 m wide. An average saturated thickness of 15 m was obtained from Figures 14 and 15. A porosity of 50% was assumed for the landfill material because landfill refuse is generally not very compact or dense. Using the above values, the volume of leachate present in the landfill is approximately $1.3 \times 10^6 \text{ m}^3$ or 1.3 billion litres.

Using a value of leachate generation of 2.3 L/sec (30 gpm), based on infiltration through the cap, the estimated annual leachate production is 7.2×10^7 litres. If all the leachate generated by infiltration through the cap were to remain in the landfill it would take 18 years just to produce the 1.3 billion litres of leachate estimated to be present in the landfill.

Groundwater Flux and Velocity

In studies of contaminated groundwater the velocity of groundwater flow is normally estimated from the relation,

$$v = Ki/n$$

where v is the average linear groundwater velocity and n is the porosity. For fractured rock n represents the percent of a unit volume of rock occupied by open fractures through which groundwater flow occurs. Larger values of n are expected to be associated with rock that has larger values of hydraulic conductivity.

In fractured rock it is very difficult to obtain accurate representative values of n . At the Upper Ottawa Street site approximate values of n were determined by several methods. The first method was the Pollard (1959) method of analysis as described in a previous section on hydraulic conductivity determinations.

The Pollard method was used on response test data where two slopes were available to calculate the fracture and matrix hydraulic conductivity and is discussed in detail in Schwartz (1975). The fracture porosity is calculated as a percentage of the total porosity of the rock, which can only be estimated. Table 7 shows the results of analyses by the Pollard method. Ranges of total porosity for shale are on the order of 0-10%, and for limestone and dolomite 0-20% (Freeze and Cherry 1979) and an average value of total

porosity (i.e. shale, 5% and limestone and dolomite 10%) was used to calculate the fracture porosity.

Results indicate that fracture porosities are generally less than 1%. For the shallow bedrock zone (i.e. cherty dolomite and upper shaly dolomite) the fracture porosity is approximately 0.12 to 0.2% with only one zone (UW10-3) considerably higher at 1.5%. Porosities in the underlying Rochester shale, determined at two locations, were .015% (UW7-3) and 0.13% (UW4-2) indicating lesser fracturing than in the upper dolomite of the Lockport Formation. The limestone and dolomite of the underlying Irondequoit and Reynales Formations and the siltstone and shale of the Thorold and Grimsby Formation showed a range of fracture porosities, from .2% to 1.3%.

Another method of determining the fracture porosity in the shallow groundwater zone for the cherty dolomite was to use the storage coefficient taken from PT3 where the shallow groundwater zone was pumped at a relatively high rate. The upper portions of this zone had storage coefficients determined, which, under unconfined conditions are a reflection of the drainable porosity, and in this case would be equivalent to the fracture porosity. Table 4 shows the values determined by the Theis and Boulton method for PT3.

Storativity values are on the order of 0.3% to 0.85% with one value of 2.7% found at UW3-2 which may be affected by high vertical leakage at this zone. Although these val-

ues of storativity may reflect a "semi-confined" condition and porosity values may be greater, the values agree well with the values determined by the Pollard method, and it is assumed the values are representative of the fracture porosity of the cherty dolomite.

Using the above porosity calculations, along with previous determinations of hydraulic conductivity and hydraulic gradient, general values for groundwater flux and velocity can be determined for the various rock units.

The horizontal linear velocity within the cherty dolomite of the Goat Island Formation can be calculated using a fracture porosity of approximately 0.2 - 0.5%, a hydraulic conductivity of 1×10^{-3} cm/sec to 1×10^{-2} cm/sec and a water table gradient of approximately 0.01 away from the influence of the groundwater mound at the landfill. The values substituted into the equation given above yield a horizontal groundwater velocity ranging from about 1.5 to 20 m/day, which are extremely high groundwater velocities.

At greater distances downgradient of the landfill (i.e. UW3, UW2, UW19) the vertical gradient in the cherty dolomite increases from being very small or negligible near the landfill to 0.25 at UW2 and 1.4 at UW19 near Albion Falls. This would indicate an increasing migration of water downward into the Rochester shale closer to Albion Falls and the Niagara escarpment. This is possibly due to stress relief of the shale, causing increased vertical fracturing nearer

the escarpment. Pump test results indicate the vertical hydraulic conductivity of the cherty dolomite is lower than the horizontal hydraulic conductivity and ranged from 5×10^{-4} to 5×10^{-6} cm/sec. The decreased hydraulic conductivity and increased vertical gradients indicate that the downward velocities in the cherty dolomite range from very small near the landfill (cm/day) to the same range as horizontal velocities (m/day) near Albion Falls and the Niagara escarpment indicating a potential for all water within the cherty dolomite to migrate downward to the shale and if significant vertical fracturing occurs within the shale, this water could potentially migrate through the shale.

Velocity calculations in the Rochester shale using fracture porosities of 0.015 - to 0.13, hydraulic conductivities from 1×10^{-9} cm/sec to 5×10^{-6} cm/sec and an average vertical gradient of 0.8 range from 0.2 m/day to 5.3×10^{-6} m/day (0.2 cm/yr). As can be seen by the wide velocity range, the determination of movement of water within the Rochester shale is very difficult because of the variability in fracturing within this unit.

The vertical velocity in the Rochester shale is an important factor in determining the potential path of leachate migration as water moves from the landfill towards Albion Falls and the escarpment in the vicinity of Albion Falls. As the vertical gradients increase in the cherty dolomite and Rochester shale with proximity to Albion Falls,

it would appear that a significant quantity of water within the cherty dolomite would either migrate along the base of the cherty dolomite and dolomite of the Gasport and Decew Formations or would migrate with depth through the Rochester shale.

Inspection of the Albion Falls area indicates a number of seeps present along an elevation equivalent to the base of the dolomite, however it does not appear that significant quantities of water move through these seeps. It should be noted that the pond located adjacent to Albion Falls at UW19 is the likely source of water for these seeps. The water in the shallow well at UW19 is a perched water table, caused by water seeping into the area from the pond. Water levels are the same for the pond and UW19-1. During drilling of UW19, once coring proceeding in the Rochester shale, water cascaded into the borehole from the upper zones for a considerable period of time.

Based on the above discussion, it would appear that a significant quantity of water in the cherty dolomite may be migrating into the Rochester shale as flow approaches the Albion Falls ravine. In order to determine if this is possible, the following calculations can be made. Using a slice of cherty dolomite, similar in length to the landfill as shown in Figure 73, which is about 700 m by 9 m and using a horizontal gradient of .01 and reasonably high hydraulic conductivity of 4×10^{-3} cm/sec the total flux through this

slice would be 2.5 L/sec, similar to the lower estimates of leachate production. This quantity of water plus the recharge from infiltration into the shaded area in Figure 3 would all have to move through the Rochester shale in order to have no seeps or springs at the base of the cherty dolomite along the Albion Falls ravine. Assuming that an annual surplus of water would be 18 cm (Gartner Lee, Associates Ltd., 1979) and that approximately half the 700 m by 500 m area would be available for infiltration (the other half being developed), this would yield a total quantity of 1 L/sec. This means that 3.5 L/sec of water must enter the Rochester shale within the shaded area of Figure 150. From the previous equation it can be seen that the bulk vertical hydraulic conductivity needed to move this water through the Rochester shale is determined by:

$$K = q/iA$$

where q is 3.5 l/sec, i is variable, ranging from .5 to 1.4 depending on proximity to the Escarpment, and assumed to be .8, and the area is approximately 700 m by 500 m. The resulting hydraulic conductivity required to allow this water to move into the Rochester shale is 1.3×10^{-6} cm/sec.

This is on the high end of the average values determined in the Rochester shale. However, a vertical hydraulic conductivity of 5×10^{-6} cm/sec was found at UW3-4 in the shale. This, plus the fact that stress relief may cause increased vertical fracturing near the Escarpment indicates

that the potential definitely exists for the water emanating as leachate from the landfill into the cherty dolomite to move completely into and through the Rochester shale by the time it migrates to the Albion Falls area. The water would eventually discharge into Redhill Creek further downstream from Albion Falls or ultimately into Lake Ontario. The quantities of potentially contaminated water relative to the volumes of water moving through the ground and being diluted prior to discharge will be very small.

CONCLUSIONS

The bedrock at the Upper Ottawa Street landfill site consists of a shallow zone (5 - 10 m thick) of moderately permeable Lockport dolomite, a zone of Rochester shale (7 - 10 m thick) beneath the dolomite and a bedded zone of limestone, shale and siltstone beneath the shale. Beneath this, at a depth of about 60 m below ground surface, there is a thick zone of Queenston shale that extends laterally to Lake Ontario.

The groundwater mound that exists within the landfill causes flow of leachate from the landfill into the shallow dolomite. The piezometer response tests and the large differentials of hydraulic head between the shallow dolomite and the strata beneath the shale indicate that the shale has a relatively low horizontal and vertical permeability. It is expected, therefore, that after leachate from the landfill enters the shallow dolomite, it is transported laterally in this rock unit by the bulk flow of groundwater. Although this lateral migration of leachate probably occurs outward from all sides of the landfill, the regional slope of the water table beyond the area of local influence of the water-table mound is generally towards Lake Ontario, probably with some convergence towards the ravine below Albion Falls. It is reasonable to expect that the regional path of

lateral leachate migration in areas away from the landfill follows this slope.

The tests conducted by pumping the open borehole (UW25) near the landfill indicate that, in the vicinity of this hole, vertical hydraulic connection through the shale is provided locally by a major zone of vertical fractures. Therefore, although the shale is generally an extensive zone of low permeability, there is a potential for downward movement of leachate to occur through it at major fracture locations such as this. Whether or not major vertical fractures exist directly beneath the landfill is not known and would be very difficult to determine.

As leachate-impacted groundwater moves eastward from the landfill in the shallow dolomite, downward leakage through the shale apparently increases considerably in zones of greater vertical fracturing in the vicinity of the steep ravine at Albion Falls. Thus, there is very little groundwater seepage evident along the cliffs near the Falls and, what seepage there is apparently originates from surface water sources or infiltration very near the ravine.

Leachate-impacted groundwater that makes its way downward through the shale in the vicinity of Albion Falls probably continues its migration towards Lake Ontario by lateral flow along more permeable bedrock strata deeper than the bottom of the ravine. In the course of this investigation no hydrogeological data were acquired from areas lakewards

of Albion Falls and therefore one can only speculate on the ultimate fate of leachate-impacted groundwater that moves downward through the shale on the landfill side of Albion Falls. It is quite possible that much or all of this water does not emerge to the surface until it passes beneath the shoreline of Lake Ontario.

Based on representative values for the eastward water table slope in the shallow dolomite in the vicinity of the landfill, on the hydraulic conductivity of this zone and on reasonable but assumed values of effective fracture porosity (0.2% - 0.5%) for this zone, calculated lateral groundwater velocities in the dolomite are about 1 to 20 m/day. At these rates contaminated groundwater could travel laterally from the landfill to the vicinity of the Albion Falls in a few years or less. It is not possible to make useful estimates of the velocity of lateral groundwater flow in the deeper strata or in the shallow bedrock beyond Albion Falls.

An estimated value of 2.3 L/s (30 gallons per minute) was obtained for the average annual rate of infiltration of water from rain and snow into the landfill. Although this estimate depends on several assumptions, it serves as an indication that the rate of leachate input from the landfill to the groundwater zone is small relative to the flow in Redhill Creek or to the drainage from other sources into Lake Ontario. It is likely that the leachate-impacted groundwater undergoes considerable dispersion as it travels in the various bedrock zones towards Lake Ontario.

The water-table mound in the landfill will persist unless infiltration of rain and snow through the cover on the landfill ceases. There has been no indication yet that the existing cover on the landfill is preventing infiltration to the degree necessary to cause the mound to decline. If an impermeable cover were to be placed on the landfill, the water-table in the landfill would decline to the bottom part of the landfill. Even under this condition, however, leachate would emanate but at a slow rate from the landfill into the permeable dolomite because lateral groundwater flow through the landfill would persist, generally from west to east.

ACKNOWLEDGEMENTS

Able assistance in the field was provided throughout the investigation by Bob Ingleton and Paul Johnson, who also constructed most of the multilevel monitoring devices. Much of the water-level monitoring and water-level response testing of piezometers was done by Vince Bulman and Roy Heiss. Dave Madge assisted in the laboratory with construction of the multilevel monitoring devices. All of the boreholes were drilled by Mike Pollice of All Terrain Drilling Limited, except for the angle hole which was drilled by Longyear Canada Limited. Water level data and stratigraphic information were compiled by Katrina Olsen and Lynn Roberts. Katrina Olsen also assisted in the evaluation of the pumping test data and of the water-level response-test data. The description of the historical development of the landfill based on interpretation of areal photographs was prepared by Lynn Roberts. The diagrams were drafted by Nadia Bahar and the manuscript was typed by Marilyn Bisgould.

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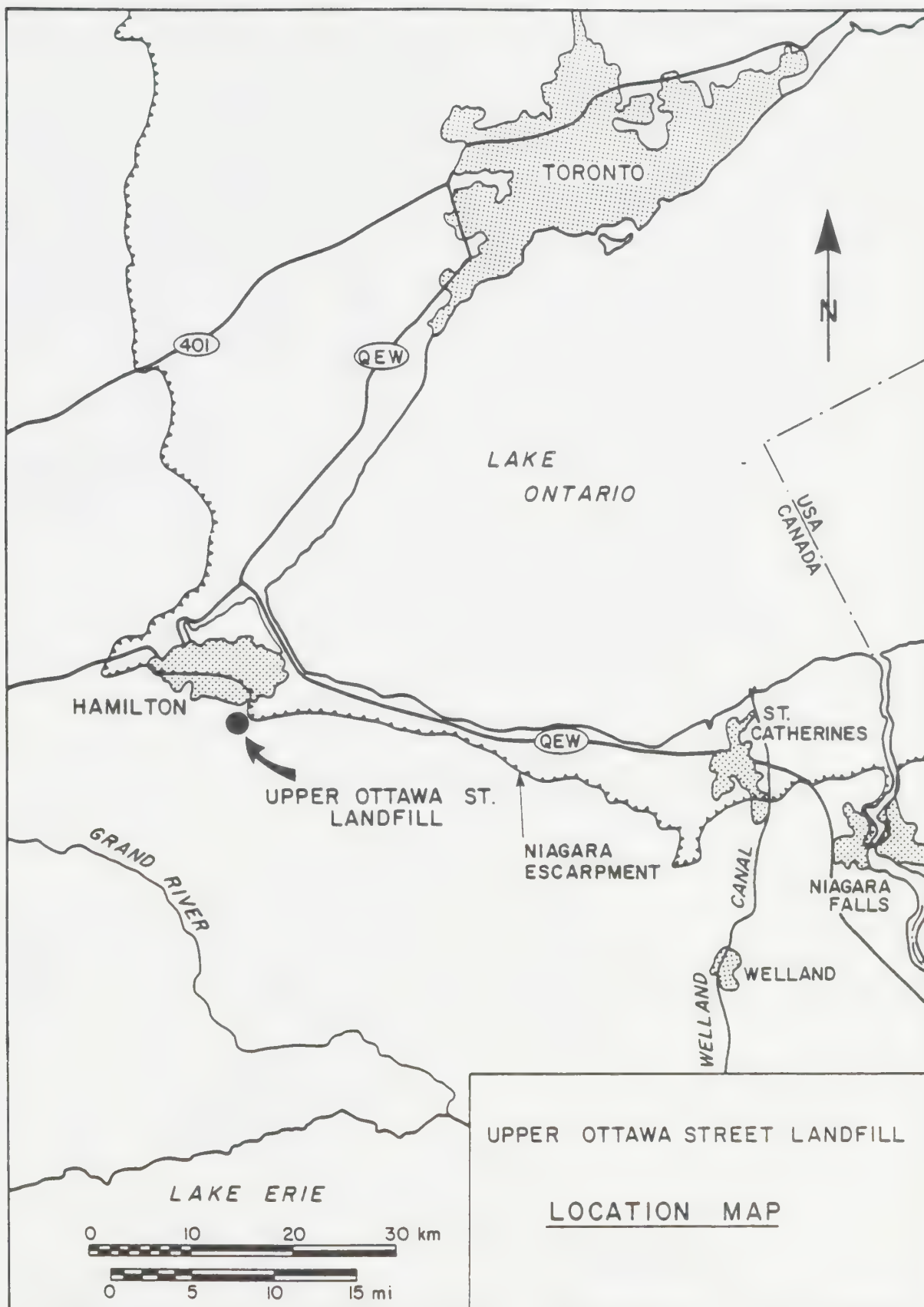


Figure 1. Location of the Upper Ottawa Street Landfill

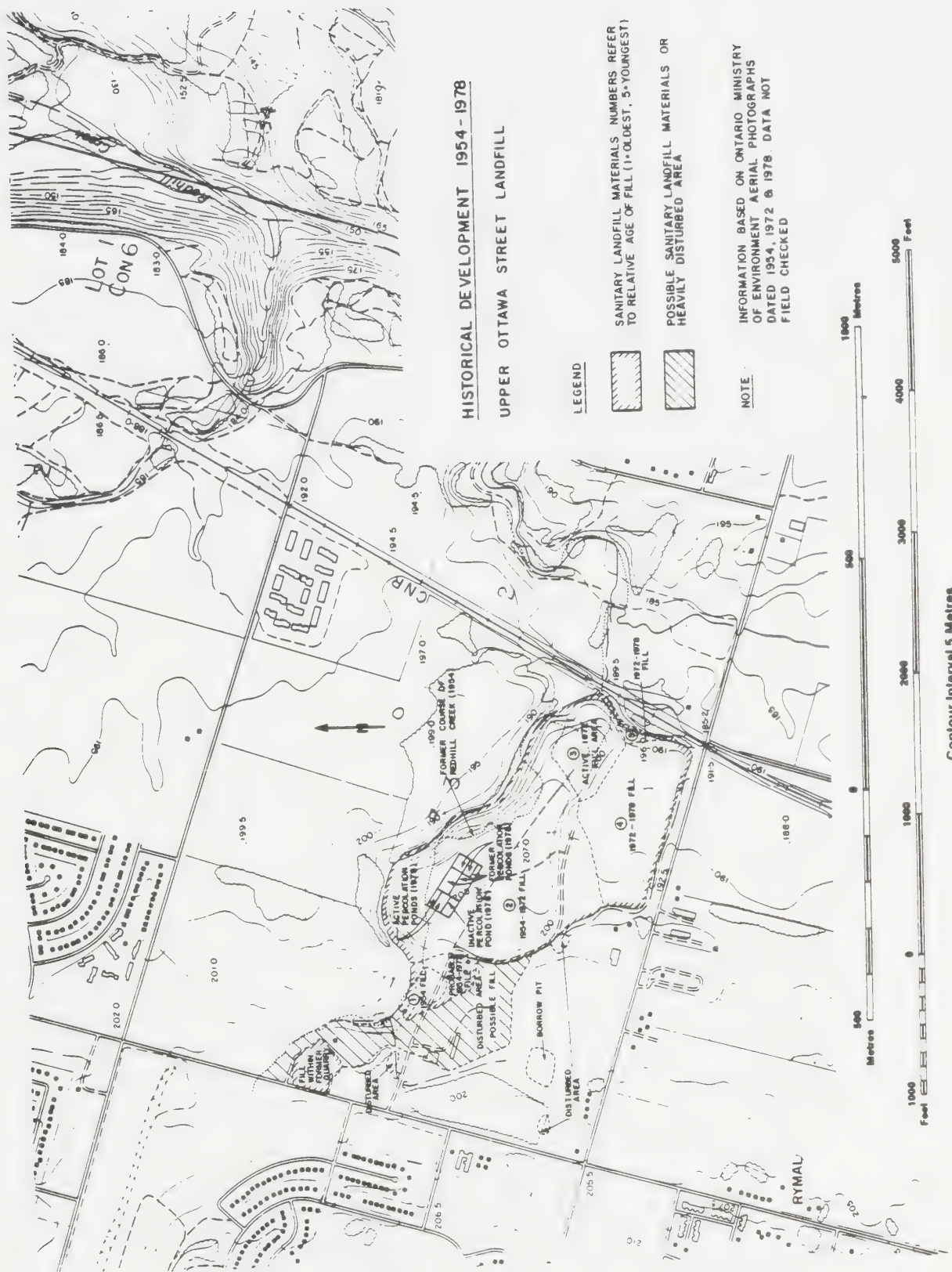


Figure 2. Historical Development

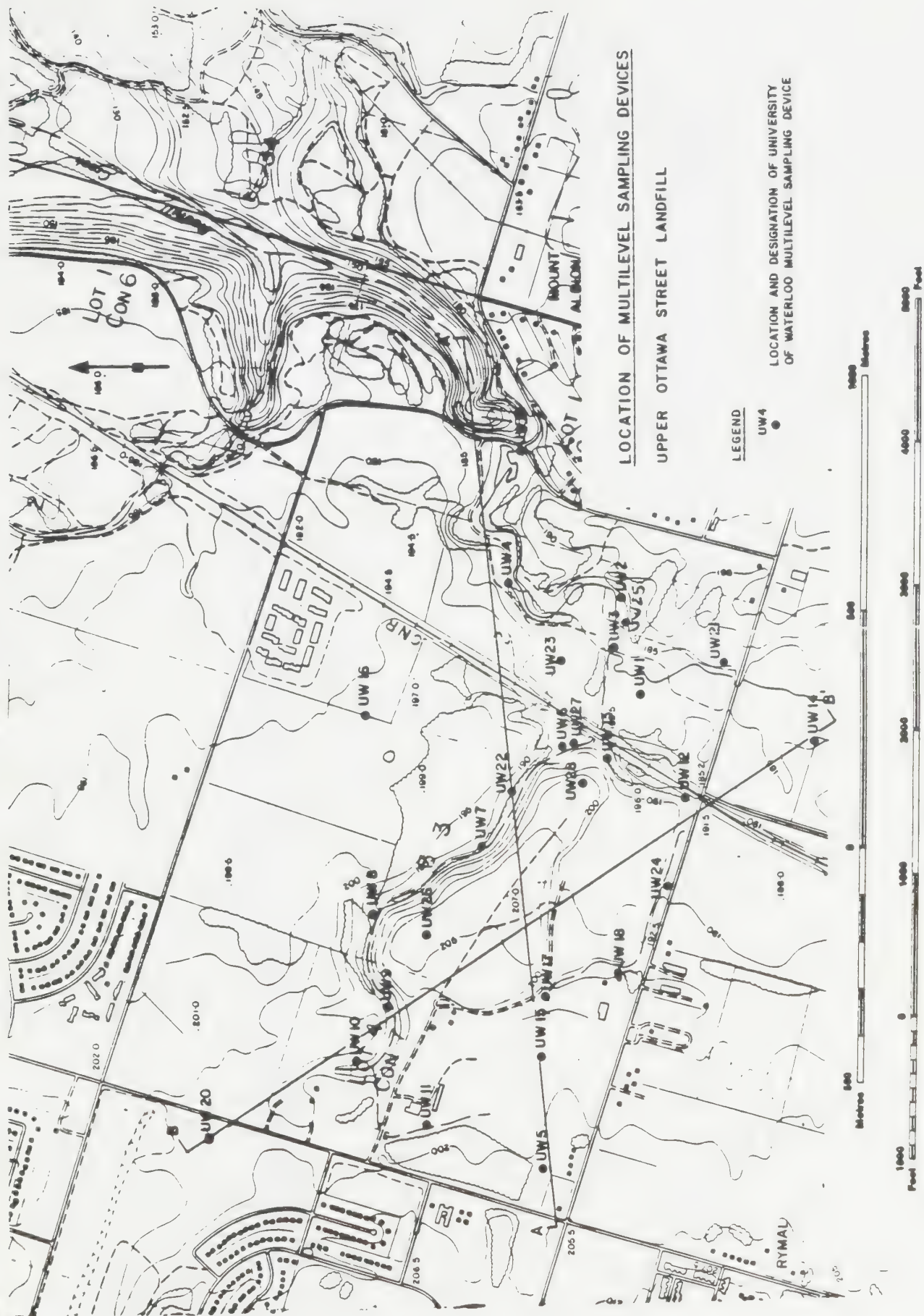
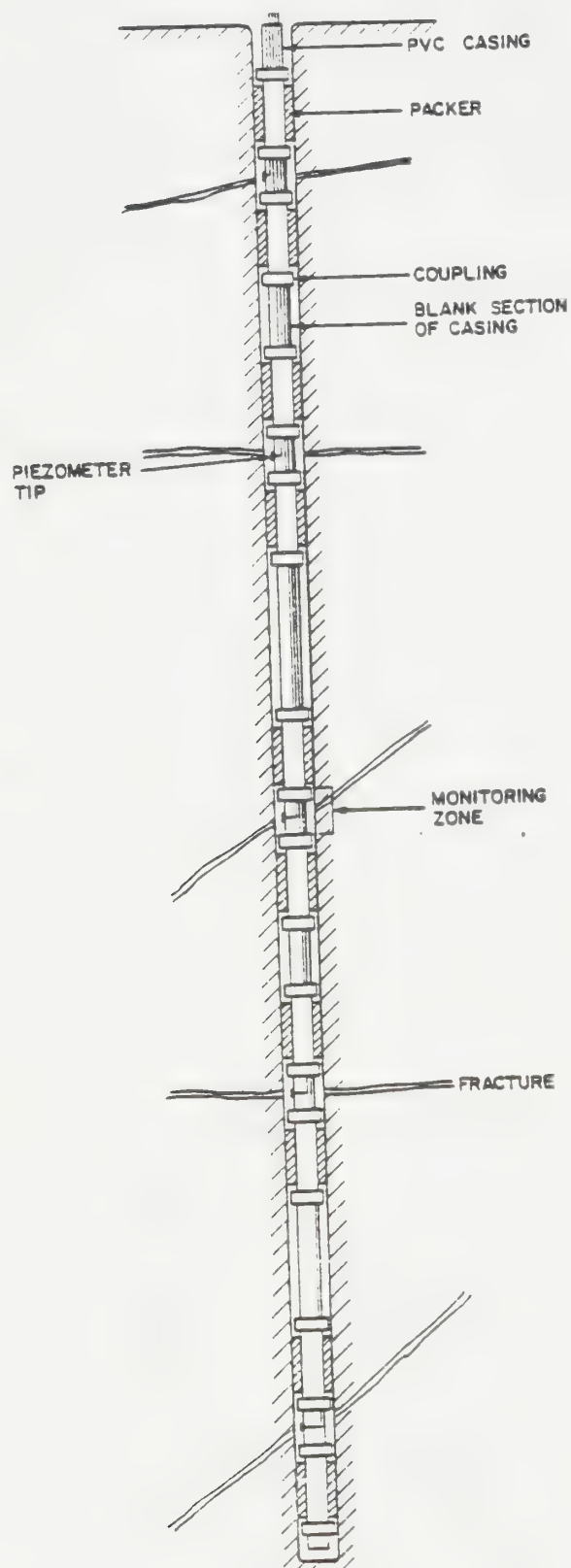
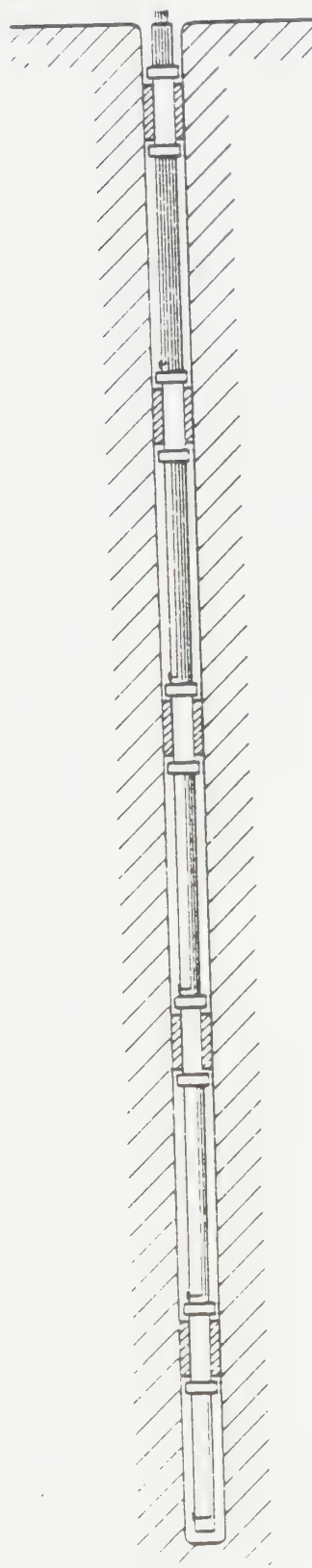


Figure 3. Locations of the multilevel monitors

DISCREET ZONE MONITORING



SEGMENTED BOREHOLE MONITORING



Schematic illustration of the multilevel device

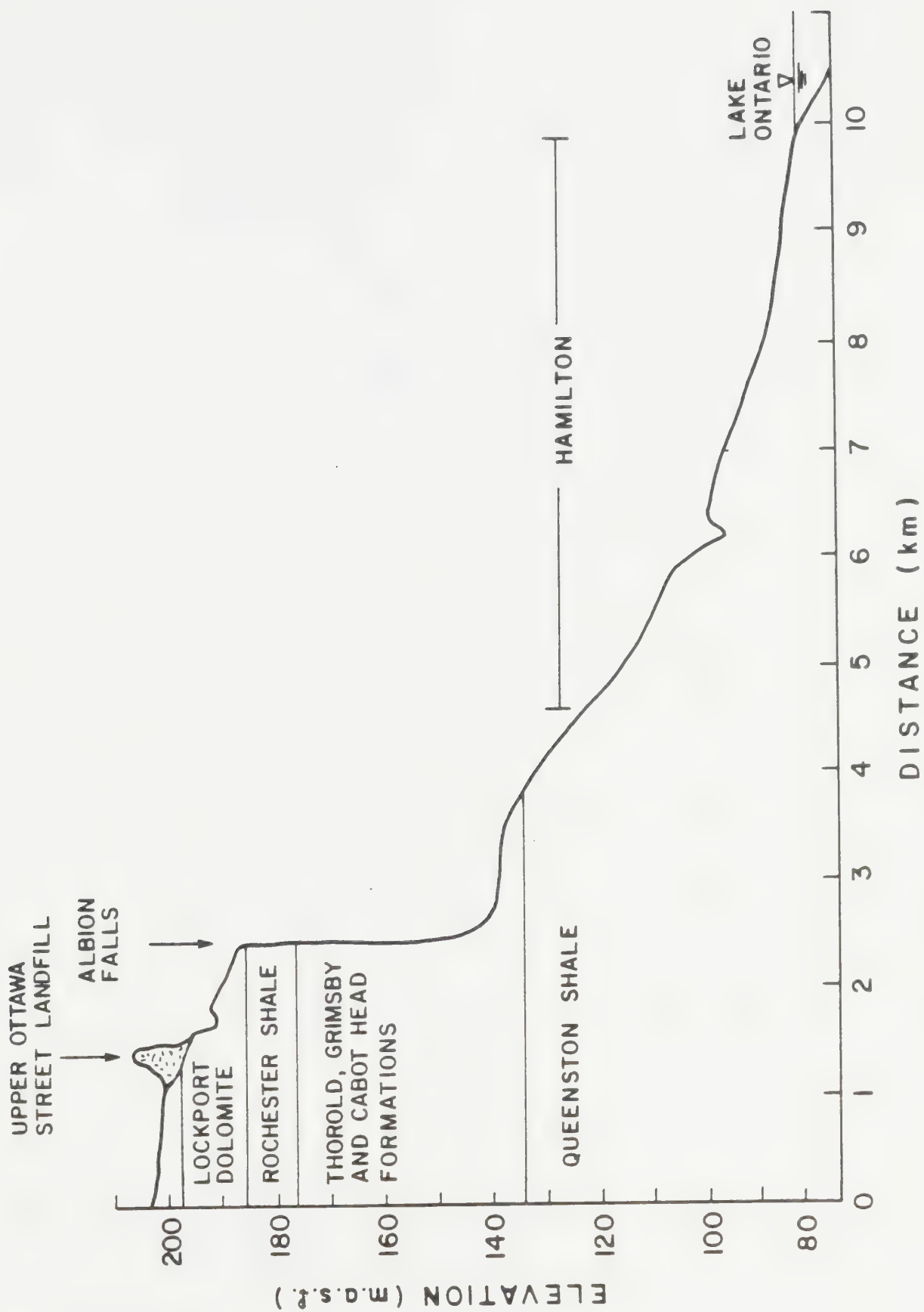


Figure 5. Regional Geologic Setting of the Landfill

West of the Landfill	East of the Landfill	Geologic Formation of Member
Clay till Dolomite to Dolomitic Shale		Eramosa
	Clay till	
Cherty Dolomite		Goat Island
Dolomite (Shaly)		Gasport
Shaly Dolomite		Decew
Shale to Shaly Dolomite		Rochester
Limestone		Irondequoit
Shaly dolomite		Reynales
Siltstone to Shale		Thorold
Red shale, Siltstone		Grimsby
Green Shale, Siltstone		Cabot Head

Figure 6. Schematic Vertical Geologic Sections
(range from 40 to 60 m thick)

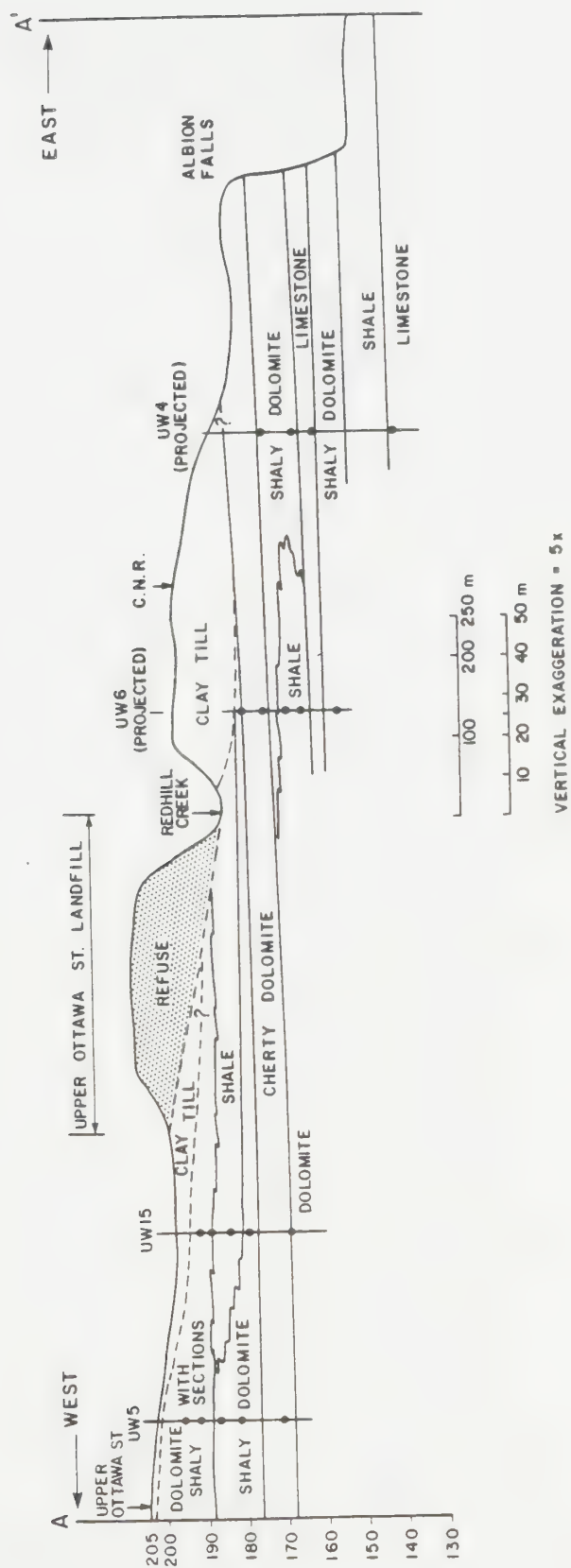


Figure 7. Geologic Cross-section A-A'

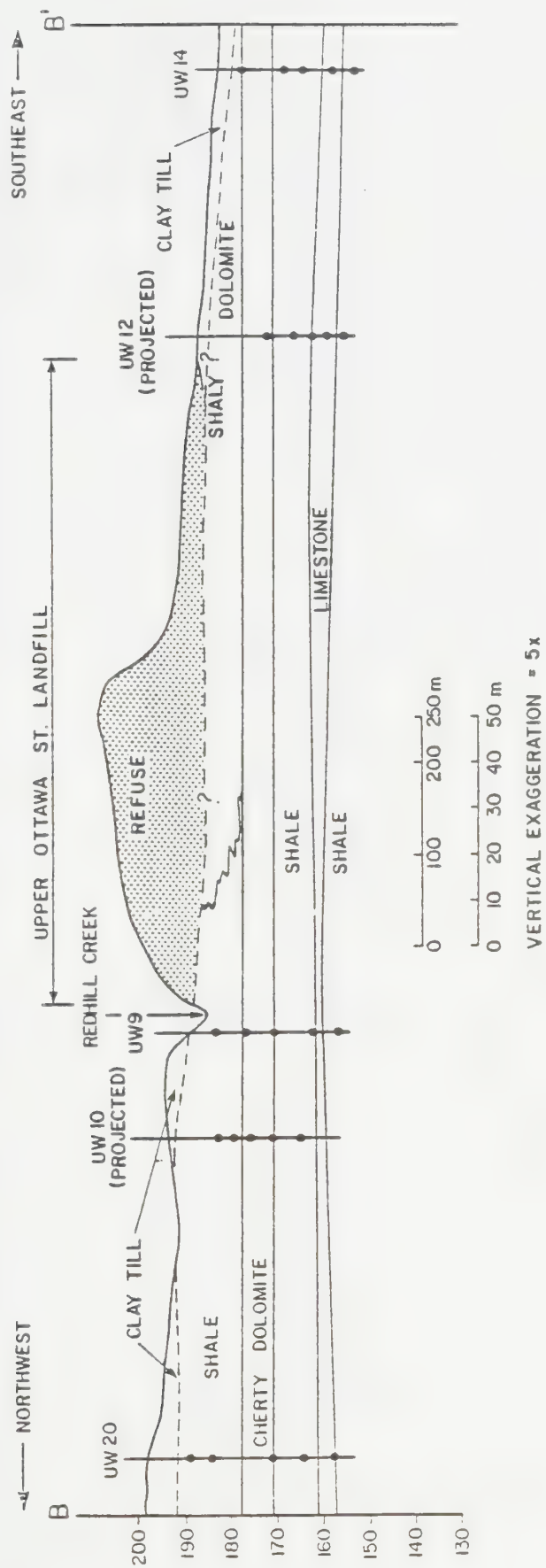


Figure 8. Geologic Cross-section B-B'

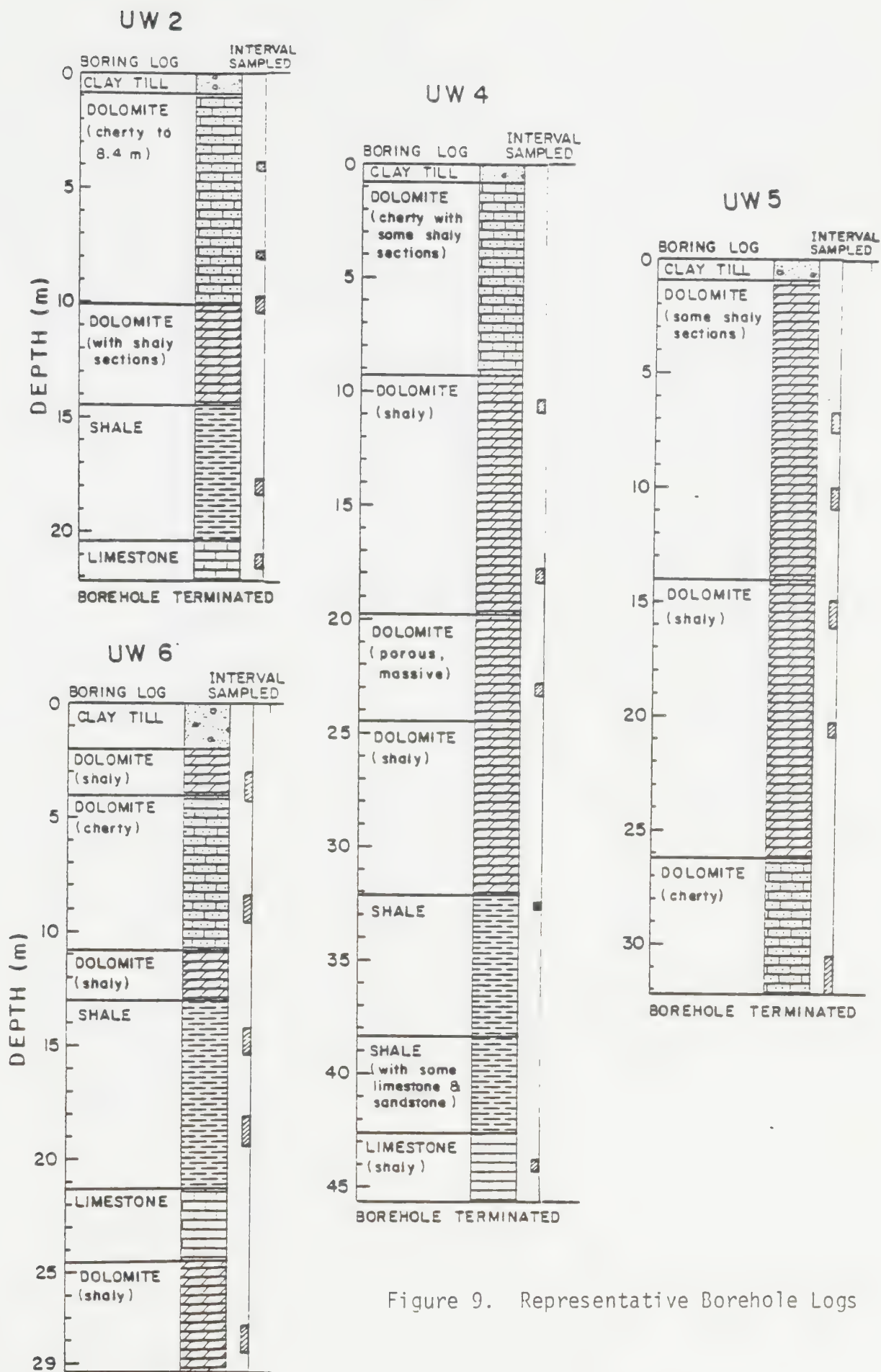


Figure 9. Representative Borehole Logs

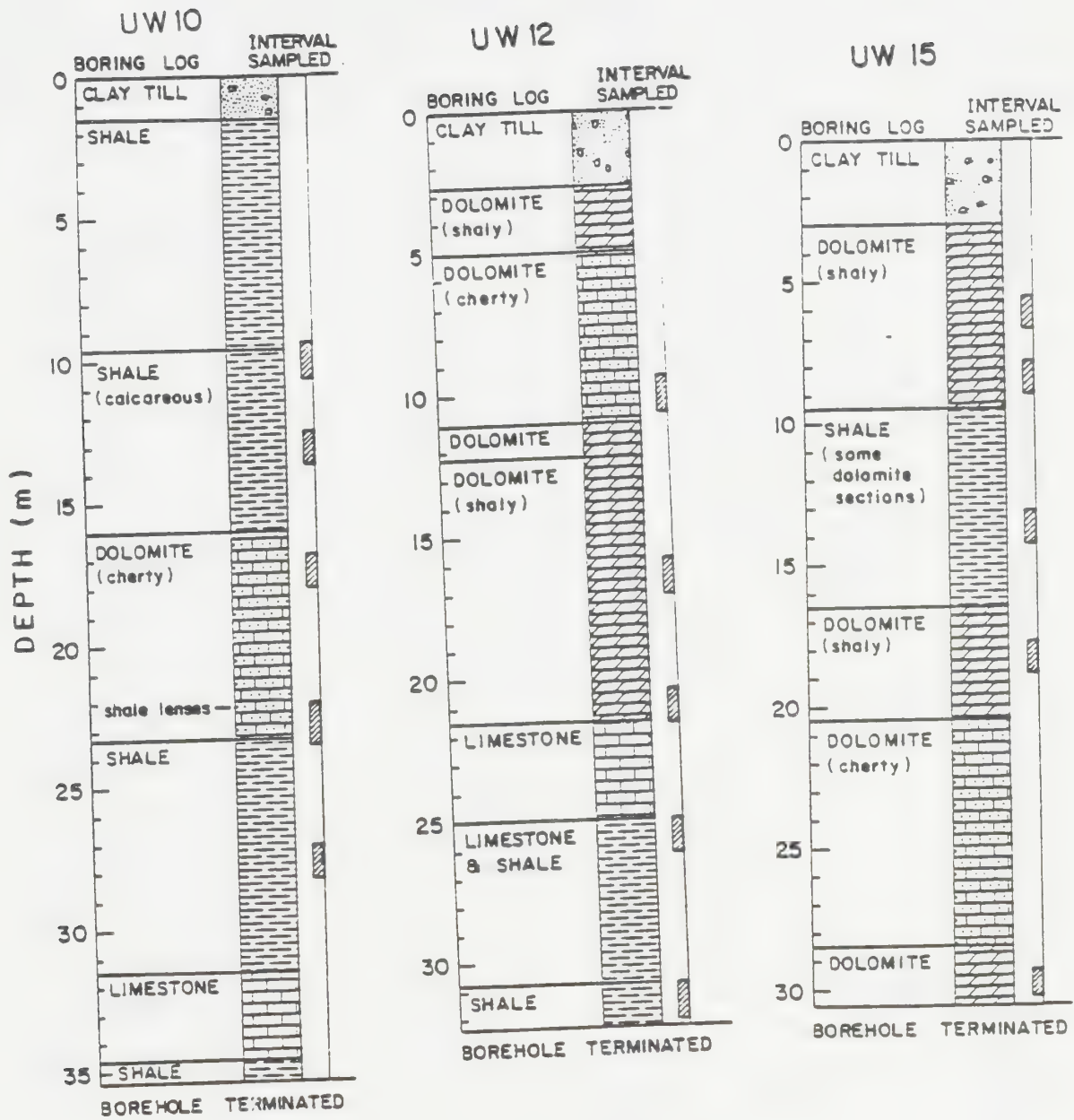


Figure 10. Representative Borehole Logs

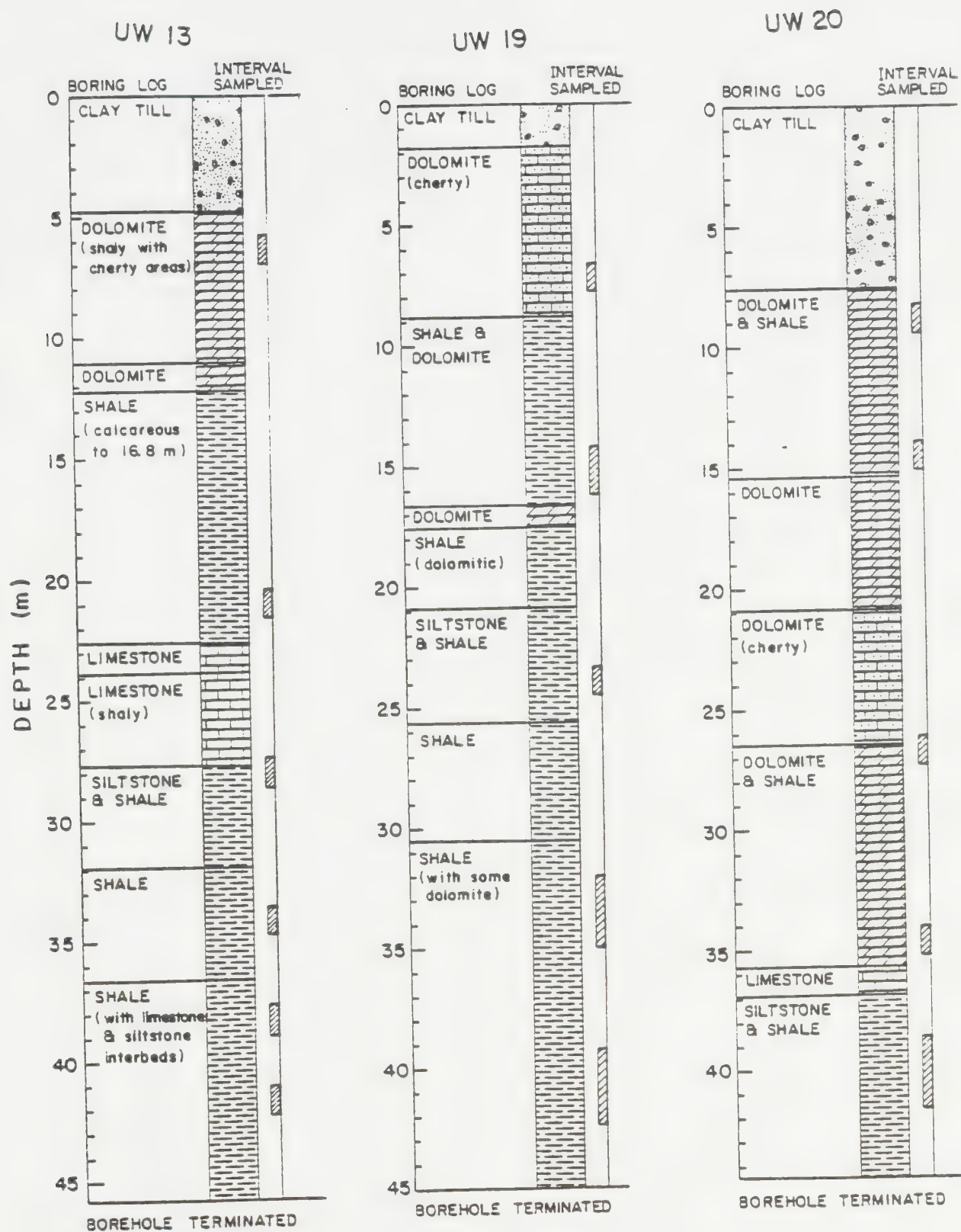
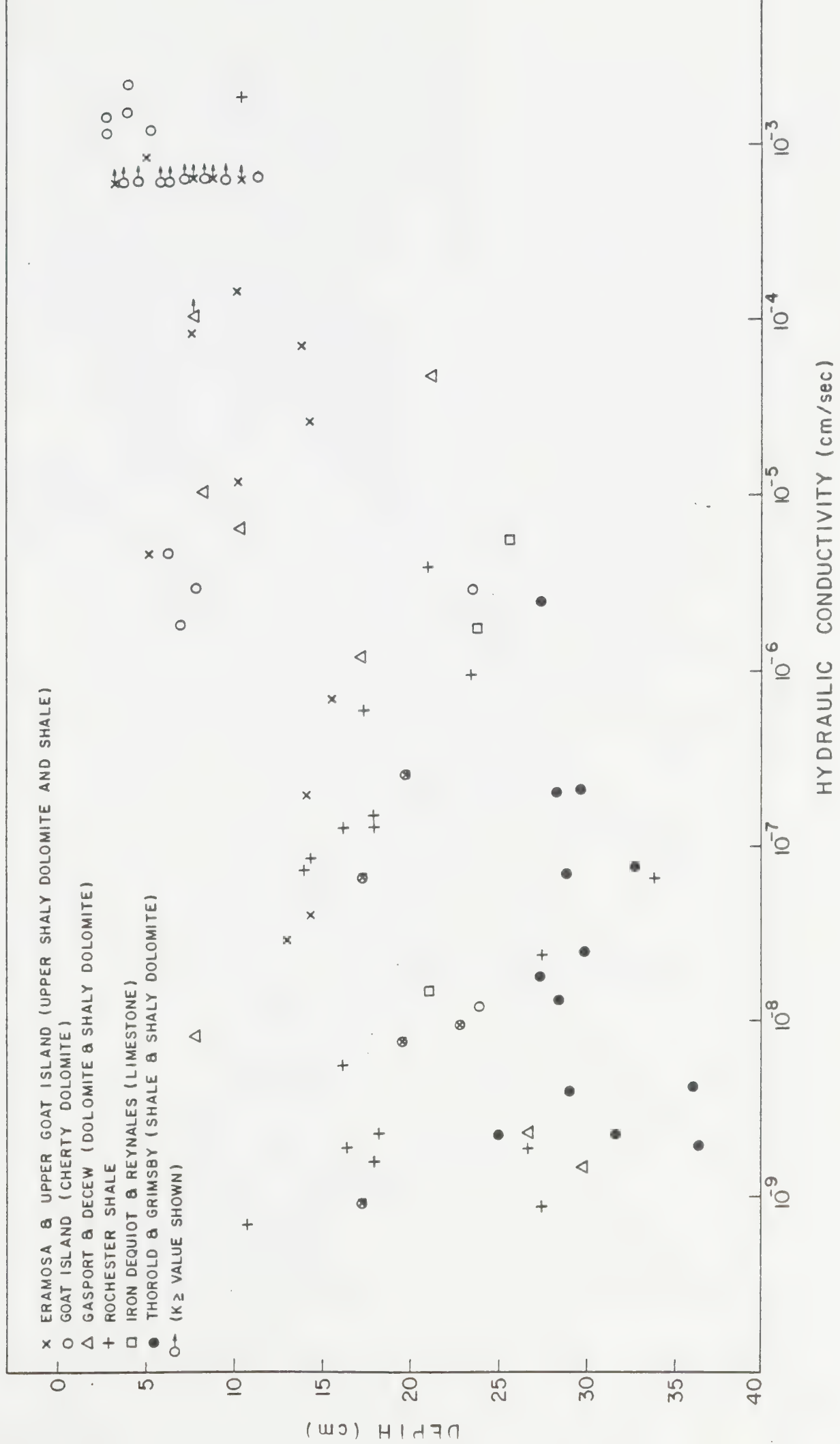


Figure 11. Representative Borehole Logs



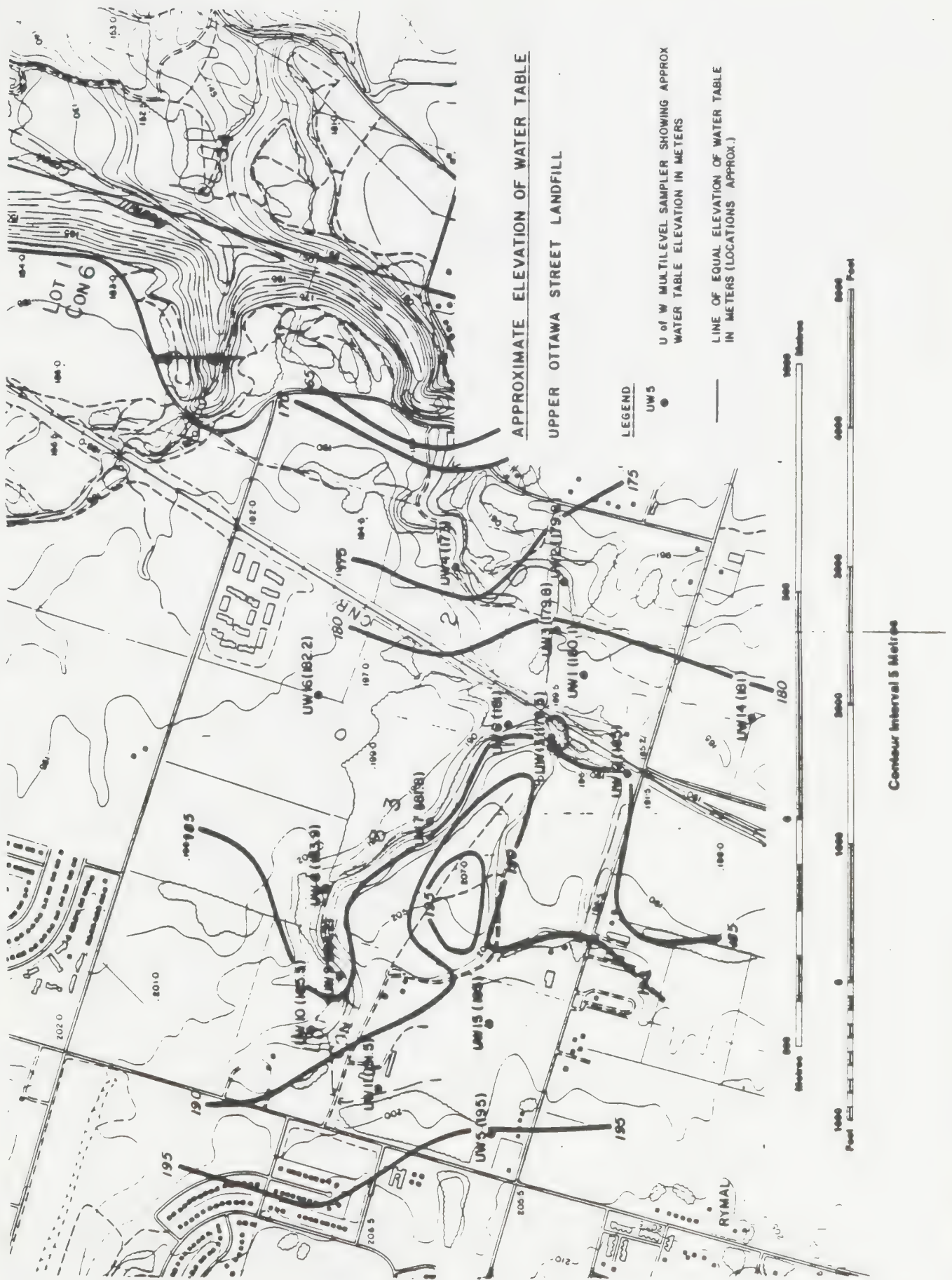
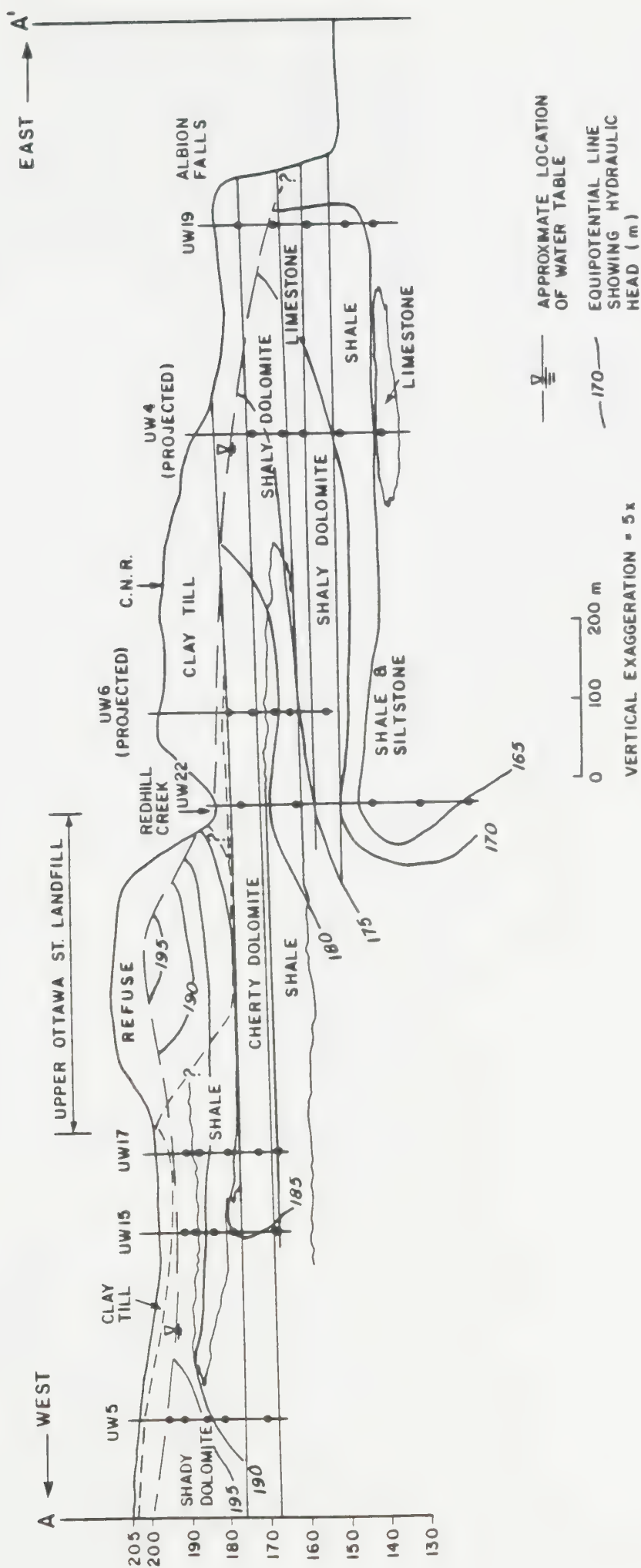
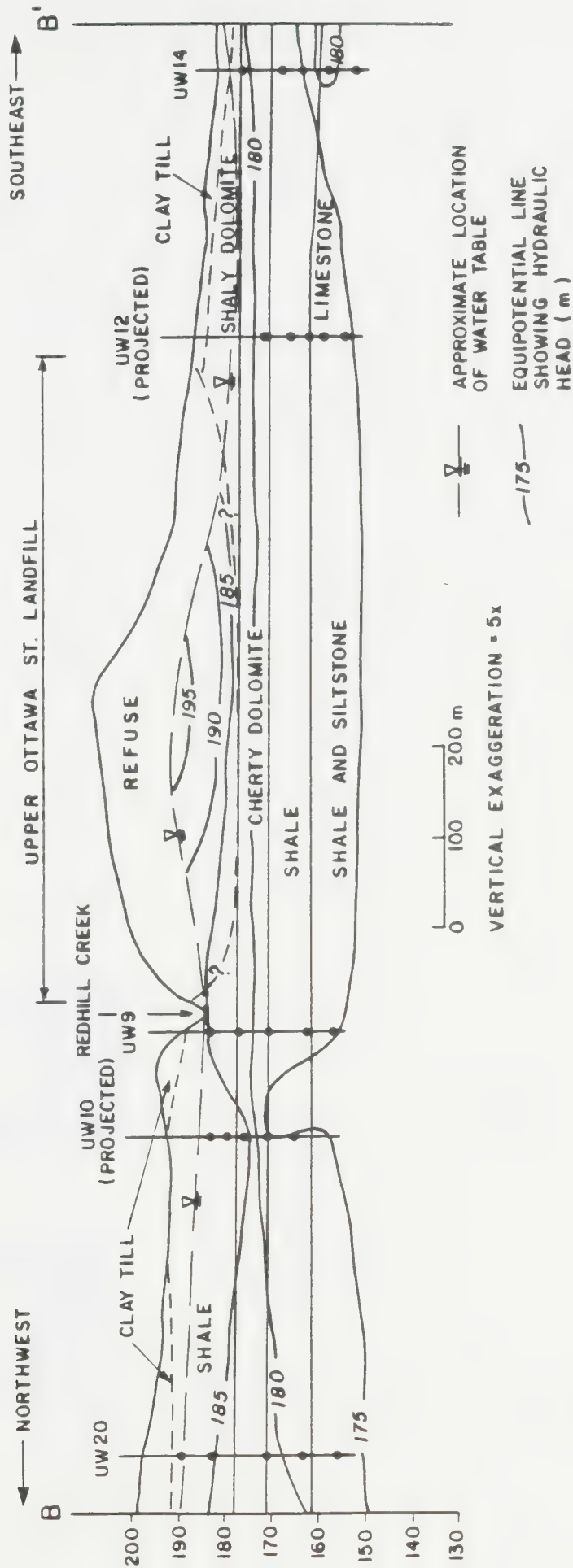


Figure 13. Water Table Map



Hydraulic Head - Cross-section A-A'



Hydraulic Head - Cross-section B-B'

Hydraulic Conductivities Determined by Response Testing

PIEZOMETER	DEPTH (m)	ELEVATION	ROCK TYPE	K ₁ (cm/sec)	K ₂ (cm/sec)
UW 1-2	4.3	178.1	cherty dolomite		$>6.1 \times 10^{-4}$
3	7.0	175.4	cherty dolomite		1.9×10^{-6}
4	8.1	174.3	cherty dolomite		$>6.1 \times 10^{-4}$
5	9.6	172.8	massive dolomite		2.3×10^{-8}
6	14.0	168.3	shaly dolomite		7.6×10^{-8}
UW 2-1	4.1	177.3	cherty dolomite	$>1.0 \times 10^{-4}$	$>9.6 \times 10^{-6}$
2	8.0	173.5	cherty dolomite		3.0×10^{-6}
3	10.1	171.3	dolomite		
4	18.1	163.3	shale		1.4×10^{-7}
5	21.3	160.1	Limestone		1.5×10^{-8}
UW 3-1	4.4	177.1	cherty dolomite (possibly massive)		$>6.1 \times 10^{-4}$
2	8.4	173.2	cherty dolomite		1.1×10^{-5}
3	10.6	171.0	shaly dolomite		6.5×10^{-6}
4	12.6	169.0	shaly dolomite		2.6×10^{-3}
5	16.3	165.3	shaly dolomite	1.2×10^{-7}	5.5×10^{-9}
UW 4-1	10.6	172.5	shaly dolomite	1.5×10^{-7}	7.9×10^{-10}
2	18.0	165.0	shaly dolomite		2.2×10^{-9}
UW 5-1	7.2	195.5	shaly dolomite		$>6.1 \times 10^{-4}$
2	10.5	192.1	shaly dolomite		$>6.1 \times 10^{-4}$
3	15.6	187.1	shaly dolomite		7.7×10^{-6}
UW 6-1	3.7	179.8	cherty dolomite		$>6.1 \times 10^{-4}$
2	9.0	174.5	cherty dolomite		$>6.1 \times 10^{-4}$
3	14.8	168.7	shale		8.0×10^{-8}
4	22.3	163.2	shale		
5	27.9	155.6	shaly dolomite	2.0×10^{-7}	1.3×10^{-8}
UW 7-1	6.4	179.1	shaly dolomite (C-S bdy)		$>6.1 \times 10^{-4}$
2	11.6	174.1	cherty dolomite		$>6.1 \times 10^{-4}$
3	17.4	168.1	shale	6.5×10^{-7}	1.6×10^{-9}
4	22.2	163.1	shale		
5	27.4	158.1	Limestone & shale	2.5×10^{-6}	1.8×10^{-8}
UW 8-2	7.3	179.3	shaly-cherty dolomite		$>6.1 \times 10^{-4}$
5	25.9	160.7	Limestone		6.1×10^{-6}
UW 9-1	3.7	184.0	shale (dolomite?)		$>6.1 \times 10^{-4}$
2	9.0	179.0	shaly-cherty dolomite		$>6.1 \times 10^{-4}$
3	17.1	170.0	shaly dolomite		1.2×10^{-6}
4	24.1	162.0	shale		
5	27.6	157.0	siltstone & shale	2.1×10^{-7}	2.5×10^{-8}
UW 10-2	13.1	180.9	shale		3.1×10^{-8}
3	17.4	176.64	cherty dolomite	6.6×10^{-8}	9.8×10^{-10}
5	27.6	166.4	shale		9.0×10^{-10}

UW 11-2	14.3	184.1	shale	2.0 x 10 ⁻⁷	4.3 x 10 ⁻⁸
3	27.1	171.3	dolomite & shale		2.3 x 10 ⁻⁸
5	44.3	154.1	siltstone & shale		4.3 x 10 ⁻⁹
					-
			shaly dolomite		1.9 x 10 ⁻⁹
			shaly dolomite		3.9 x 10 ⁻⁶
			Limestone & shale		2.2 x 10 ⁻⁹
			shale		2.2 x 10 ⁻⁹
			shale		4.9 x 10 ⁻⁵
UW 13-2	21.0	163.9	shale (with Is)		1.4 x 10 ⁻⁶
5	38.3	146.7	shale		2.0 x 10 ⁻⁸
6	41.6	143.4	shaly-cherty		>6.1 x 10 ⁻⁴
UW 14-1	5.9	177.4	dolomite		
4	24.1	159.3	Limestone	1.7 x 10 ⁻⁶	2.8 x 10 ⁻⁷
5	29.1	154.2	siltstone & shale	7.0 x 10 ⁻⁸	3.8 x 10 ⁻⁹
UW 15-1	6.1	193.4	shaly dolomite		>6.1 x 10 ⁻⁴
2	8.5	191.0	shaly dolomite		>6.1 x 10 ⁻⁴
4	18.3	181.3	shaly dolomite		
5	29.7	169.8	dolomite		1.5 x 10 ⁻⁹
UW 17-1	4.6	193.1	shaly dolomite		>1.8 x 10 ⁻⁵
2	8.7	188.9	shale		>1.8 x 10 ⁻⁵
4	22.9	174.8	cherty dolomite		9.4 x 10 ⁻⁹
5	26.8	170.8	dolomite		2.4 x 10 ⁻⁹
UW 18-1	5.2	188.2	shale		4.9 x 10 ⁻⁶
2	10.7	182.7	shale		>6.0 x 10 ⁻⁵
3	14.0	179.4	dolomite		7.1 x 10 ⁻⁵
4	19.8	173.6	cherty dolomite	2.5 x 10 ⁻⁷	7.8 x 10 ⁻⁹
5	27.1	166.3			
UW 19-1	7.2	173.1	cherty dolomite		
2	15.2	165.1	shaly dolomite		-
UW 20-3	26.7	173.3	shaly dolomite		1.3 x 10 ⁻⁹
			(and D & S)		
4	34.6	165.4	dolomite & shale		6.6 x 10 ⁻⁸
5	40.1	159.9	siltstone & shale		1.4 x 10 ⁻⁹
UW 21-3	23.5	162.6	shale		9.8 x 10 ⁻⁷
5	32.9	153.2	siltstone (shale)		8.0 x 10 ⁻⁷
				2.7 x 10 ⁻⁶	2 x 10 ⁻⁷
UW 25-II-1	1.7-2.9	179.1	cherty dolomite		1.2 x 10 ⁻³
II-2	3.4-4.6	177.4	cherty dolomite		1.6 x 10 ⁻³
II-3	5.0-6.2	175.8	cherty dolomite		1.2 x 10 ⁻³
25-III-1	1.6-2.8	179.2	cherty dolomite		1.4 x 10 ⁻³
III-2	3.7-4.9	177.1	cherty dolomite		2.2 x 10 ⁻³
III-3	5.8-7.0	175.0	cherty dolomite		1.6 x 10 ⁻⁴

Table 2. Water level Fluctuations

WELL	DEPTH (metres)	ROCK TYPE	ROCK UNIT	WATER LEVEL RISE (metres)	WATER LEVEL DECLINE (metres)	DATE OF FLUCTUATIONS (1983)
UW 1-2	4.3	cherty dolomite	Goat Island	.78	1.09	01/02 28/07
UW 1-3	7.0	cherty dolomite	Goat Island	.68	0.88	01/02 28/07
UW 1-4	8.1	cherty dolomite	Goat Island	.67	0.92	01/02 28/07
UW 1-5	9.6	dolomite	Gasport	1.74	-	14/02 25/05
UW 2-2	8.0	cherty dolomite	Goat Island	.41	.61	01/02 28/07
UW 2-3	10.1	dolomite	Gasport	.40	.62	01/02 28/07
UW 2-4	18.1	shale	Rochester	1.28	-	10/12/82 -
UW 2-5	21.3	Limestone	Irondequoit	4.65	-	14/02 -
UW 3-2	8.4	shaly dolomite	Rochester	.46	.62	01/02 28/07
UW 3-3	10.6	shaly dolomite	Rochester	.63	1.06	01/02 28/07
UW 3-4	12.6	shaly dolomite	Rochester	.19	.81	01/02 28/07
UW 5-1	7.2	shaly dolomite	Eramosa	1.58	1.95	01/02 28/07
UW 5-2	10.5	shaly dolomite	Eramosa	1.57	1.85	01/02 28/07
UW 5-3	15.6	shaly dolomite	Eramosa	-	.26	13/04 28/07
UW 6-1	3.7	cherty dolomite	Goat Island	.48	.58	15/02 27/07
UW 6-2	9.0	cherty dolomite	Goat Island	.56	.91	15/02 27/07
UW 6-3	14.8	shale	Rochester	.15	1.65	24/01 11/05
UW 6-5	27.9	shaly dolomite	Thorold	2.38	3.66	07/02 15/02
UW 7-1	6.4	shaly dolomite	Eramosa	.42	.77	15/02 27/07
UW 7-2	11.6	cherty dolomite	Goat Island	.47	.81	15/02 27/07
UW 7-3	17.4	shale	Rochester	.18	2.9	15/02 27/07
UW 7-4	22.3	shale	Rochester	-	±.20	- -
UW 7-5	27.4	Limestone	Irondequoit	-	±.25	- -
UW 8-2	7.3	dolomite	Eramosa	.34	.70	15/02 27/07
UW 8-3	13.2	cherty dolomite	Goat Island	.29	.42	15/02 27/07
UW 8-4	20	shale	Rochester	-	±.15	- -
UW 8-5	24.9	Limestone	Irondequoit	1.67	.92	- -
UW 9-1	3.7	shale	Eramosa	-	±.25	- -
UW 9-2	9.0	cherty dolomite	Goat Island	.31	.66	15/02 27/07
UW 9-4	24.1	shale	Rochester	-	±.30	- -
UW 10-1	10.0	shale	Eramosa	.34	.32	15/02 13/07

Table 2 (continued)

Table 3. Vertical Connectivity

	Piezometer				
<u>UW1</u>	<u>2</u>	<u>3</u>	<u>4</u>	<u>5</u>	<u>6</u>
Response K	$>6.1 \times 10^{-4}$	1.9×10^{-6}	$>6.1 \times 10^{-4}$	2.3×10^{-8}	7.6×10^{-8}
(cm/sec.)					
Water level	180.23	180.09	180.21	179.98	182.36
(m.a.s.l.)					
	0.02	TT-975ml	0.09	-	-
	PP-22 L	0.01	0.00	-	-
		-	TT-475ml	0.00	0.00
<u>UW2</u>	<u>II</u>	<u>2</u>	<u>3</u>	<u>4</u>	<u>5</u>
Response K	-	1.0×10^{-4}	$>6.1 \times 10^{-4}$	1.4×10^{-7}	1.5×10^{-8}
(cm/sec.)					
Water level	180.37	179.08	179.00	I	I
(m.a.s.l.)					
	0.01	PP-50 L	0.07		
<u>UW3</u>	<u>II</u>	<u>2</u>	<u>3</u>	<u>4</u>	<u>5</u>
Response K	-	1.1×10^{-5}	5.5×10^{-6}	2.6×10^{-3}	1.2×10^{-7}
(cm/sec.)					
Water level	180.10	179.51	179.12	179.55	178.29
(m.a.s.l.)					
	0.00	PP-2 L	0.37	0.00	-
	-	0.07	0.00	TT-9 L	I
	-	0.04	TT-850ml	0.00	I
	-	-	0.00	0.00	TT-850ml
<u>UW21</u>	<u>1</u>	<u>2</u>	<u>3</u>	<u>4</u>	<u>5</u>
Response K	C	C	9.8×10^{-7}	-	8.0×10^{-7}
(cm/sec.)					
Water level	180.12	180.31	176.07	174.67	173.62
(m.a.s.l.)					
August 19	-	0.00	0.00	TT-450ml	0.00
	TT-100ml	0.00	0.00	-	-
	-	0.00	TT-50ml	I	0.00
August 22	0.00	0.00	TT-1300ml	0.00	-
	-	0.00	I	0.00	TT-2300ml
	0.00	0.00	I	TT-240ml	I
	0.00	PP-200ml	I	I	I

KZY

TT = water removed by a triple tube sampler

PP = water removed by a peristaltic pump

C = pumpout has been noted to be continuous in early summer

I = previous sampling has affected the water level

- = no measurement

K = hydraulic conductivity

Table 4. Porosity Values

WELL	ROCK TYPE	DEPTH (m)	POLLARD % of total porosity	POLLARD % of total volume	THEIS % of total volume	BOULTON % of total volume
UW 2-II	Goat Island Dolomite	3.0	-	-	0.32	-
UW 2-2	Goat Island Dolomite	8.0	2.0	0.2	0.3	-
UW 2-3	Goat Island Dolomite	10.1	-	-	0.46	-
UW 3-II	Goat Island Dolomite	3.0	-	-	0.85	-
UW 3-2	Goat Island Dolomite	8.4	-	-	-	2.7
UW 4-2	Rochester Shale	18.0	2.6	0.13	-	-
UW 6-5	Thorold Shale and Siltstone	27.9	8.6	0.86	-	-
UW 7-3	Rochester Shale	17.4	0.3	0.015	-	-
UW 7-5	Irondequoit Limestone	27.4	2.0	0.2	-	-
UW 9-5	Thorold Shale and Siltstone	29.9	12.8	1.3	-	-
UW 10-3	Goat Island Dolomite	17.4	15.0	1.5	-	-
UW 11-2	Eramosa Shale	14.3	2.4	0.12	-	-

PART 2

GROUNDWATER CHEMISTRY AT THE UPPER OTTAWA STREET
LANDFILL SITE

INTRODUCTION

The principal objective of the chemical sampling program at the Upper Ottawa Street Landfill was to determine whether the landfill is a source for any chemical species detected in the groundwater within the vicinity of the site. In order to identify landfill leachate impact, representative groundwater must be obtained, reliable and comprehensive analyses conducted, and the results interpreted in light of hydrogeological information.

In order to maximize the chance of observing any adverse chemical effect that the landfill might be having upon the groundwater, the sampling and analyses encompassed a broad range of chemical parameters. Studies conducted at other landfills have demonstrated that relative to "background" conditions, groundwater contaminated by landfill leachate may exhibit elevated temperatures (MacFarlane et al., 1983); lower levels of nitrate and dissolved oxygen (Baedecker and Back, 1979), and elevated levels of major cations, major anions, reduced nitrogen species, heavy metals, methane and total organic carbon (Robinson and Maris, 1979; Nicholson et al., 1983; Cherry, 1983; Baedecker and Back, 1979). Consequently, all of these parameters were included in the chemical sampling and analysis program at the Upper Ottawa Street site. Analyses for tritium (^3H)

were used to draw inferences regarding the age (time since recharge) of the groundwater. Both purgeable and extractable organic compounds were also incorporated into the program, since preliminary work at this site suggested that these compounds - particularly the purgeable organics - were likely to be among the most reliable indicators of groundwater contamination by the Upper Ottawa Street landfill. Additionally, samples from selected points were analysed for pesticides and polychlorinated biphenyls (PCBs), organic species which may be present in leachate generated from industrial wastes.

GROUNDWATER SAMPLING AND ANALYSIS

Piezometer Flushing

There is considerable uncertainty about the most appropriate piezometer flushing and sampling protocol for obtaining groundwater samples. In deposits such as sand or gravel, removal of three to eight well volumes is thought to be an appropriate compromise between flushing the water standing in the piezometer and drawing groundwater in from more distant locations in the aquifer. In low-matrix-permeability, highly-fractured rocks, this flushing may draw in distant groundwater and the sample would not be representative of the immediate vicinity of the piezometer point. A project was undertaken (McKee, 1983) to evaluate the changes in groundwater quality during the removal of up to twelve standing piezometer volumes from UW9-1 and UW10-1. A peristaltic pump drew water to the surface where electrical conductivity, pH and Eh were measured. Samples were returned to Waterloo and to the MOE laboratory in Toronto for chemical analyses. Results are shown in Figure 1 where selected parameters are plotted versus the number of well volumes removed prior to the particular sampling. Well volumes for UW9-1 and UW10-1 are 3.14 l and 3.24 l, respectively.

For UW9-1, chloride is the dominant anion and so a good correlation between specific conductance (electrical conductivity) and chloride was anticipated. Figure 1 indicates a poor correlation up to about 1 well volume and then a fair correlation up to about 8 well volumes. The TOC concentration is extremely variable, while pH seems to stabilize to about 6.80 after about 5.5 volumes. For piezometer UW9-1, stability for all but pH and Mg does not seem to have been reached within 12 well volumes of flushing.

Some additional parameters are included in Figure 1 for UW10-1. Changes in the bulk chemistry of the sampled groundwater are evident. Once again, TOC variation differs significantly from that of either chloride or specific conductance. UW10-1 showed relatively stable pH values at 6.80 after 3 well volumes. Eh was rather unstable with values ranging considerably about a possible mean of +170 (mV) after about 1 well volume was removed. The concentration of iron, an Eh and pH sensitive parameter, was also unstable. After about 1 well volume was pumped, the iron trend was somewhat sympathetic to the Eh trend (Figure 1). This similarity was not expected since iron is usually found at higher concentrations in groundwaters with lower Eh values (i.e. more reduced waters). Magnesium concentrations are more uniform at about 62 mg/l.

Although these studies are of limited scope, the following observations are made:

1. Few parameters co-vary. If stability in a chemical parameter is taken as indicative of a representative groundwater being obtained, then stability of the specific parameter(s) of interest must be confirmed.
2. Large variations are apparent during initial (< 1 well volume) water withdrawal and are probably due to flushing of stagnant water from the sampling tube. Significant but smaller variations occur for most parameters up to 12 well volumes. These could be due to mixing of chemically-different groundwaters due to overpumping. Removal of large volumes of groundwater will therefore not provide stabilization nor a representative sample.
3. Flushing of perhaps two or three well volumes seems reasonable and practical.

At the Upper Ottawa Street landfill site, all piezometers had been purged of some or all water standing in the piezometer on a number of occasions before final sampling. From 0.1 to 2 standing volumes were removed immediately prior to sample collection with the lower volumes removed from piezometers with slow recharge rates (< 100 cm³ per hour). A complete suite of samples from a piezometer was collected within 1.5 days and so when recharge was slow, only minimal flushing was possible and insufficient water for measurement of all parameters was obtained.

Sampling Procedures.

The chemical sampling involved a high regard for sample integrity and reliability, coupled with a recognition of the practical constraints imposed by field conditions. The sampling protocol was made sufficiently flexible to accommodate either rapidly-or slowly-recharging points. A peristaltic pump was used to obtain samples when the depth to water was less than approximately 8 m. When the depth to water exceeded 8 m, a gas-drive, or "triple-tube" sampler (Robin et al., 1982) was used.

Appendix A summarizes the sampling sequence. The methods by which the different types of samples were obtained and preserved, as well as the procedures used to clean the sample containers, are outlined following the flow chart.

Total organic carbon (TOC) and chloride (Cl) analyses served several different functions during this study. "TOC/Cl" samples were taken during routine piezometer flushing operations until levels of both parameters stabilized, providing a rough indication that most of the drill water had been removed. During the chemical sampling program, TOC/Cl samples were taken at the beginning and end of each set of samples, and were used to indicate any major changes in groundwater chemistry which may have occurred over the course of an individual sampling episode. In addition, the relatively high TOC/Cl sampling frequency made it possible to monitor long-term changes in these two parameters at individual points during the two year sampling program.

Since the purgeable and extractable organic compounds were anticipated to be among the most likely indicators of landfill contamination, samples for these parameters were taken immediately following the initial TOC/Cl samples. After field measurements of groundwater temperature, pH and conductivity were made, filtered samples for major inorganic parameters, nitrogen species and heavy metals were taken, followed in turn by samples for dissolved gases, environmental isotopes and TOC/Cl.

Analytical Methods and Results.

Appendix B summarizes the methods used to chemically analyse all of the groundwater samples taken during the Upper Ottawa Street Landfill Study. Analyses were performed by commercial, provincial government and university laboratories with sufficient quality assurance checks, in our opinion, to provide reliable data.

The groundwater analyses for "routine" parameters are presented in Appendix C and include results for inorganics, dissolved gases, purgeable organics and aqueous isotopes. The more difficult analyses for trace organics were conducted on far-fewer samples. The analyses are discussed in a later section and the results are summarized in Tables 8, 9 and 10. Additional trace organic analyses performed by the Ontario Ministry of the Environment laboratories are more fully reported in Appendix D. Analyses performed by Stanford University are also more fully reported in Appendix E.

The pertinent results from these sources are included in Tables 8, 9 and 10.

QUALITY CONTROL PROCEDURES.

Table 1 lists the procedures employed on a weekly basis during the sampling program to check on the reliability of the analytical results. Water for sampling and procedural blanks was obtained from a carbon filtration/ultraviolet irradiation unit for organic compound analyses and from a glass distillation apparatus for the analyses of inorganic species. Sample replication on a weekly basis was employed only for samples taken for the analysis of major ions/nitrogen species, heavy metals and some aqueous isotopes. Sufficient replication for all of the remaining parameters (i.e. organic compounds and dissolved gases) had already been incorporated into the regular sampling procedure.

In addition to the routine weekly analysis of duplicate samples for purgeable organics, 11 samples were obtained in quadruplicate during the period from October 25 to November 22, 1983. These were taken for inter-laboratory comparison between Mann Testing Laboratories (MTL) using GC/MS and the Organic Geochemistry Laboratory at the University of Waterloo using GC only, with two replicates from each sample going to each laboratory. Six of these were analysed by MTL using a chromatographic column identical to that used by the laboratory at Waterloo (a DB-5 fused silica column) in order to confirm the peak identifications provided by the Waterloo

laboratory and to check for the presence of co-eluting species. The remaining five samples were analysed by MTL using a different column (a Superox fused silica column) to provide a comprehensive GC/MS inventory of all compounds which could be identified in those samples.

QUALITY CONTROL RESULTS.

BLANKS.

Tables 2, 3 and 4 document the levels of inorganic and purgeable organic species found in blank samples analysed during the course of the study. The sampling equipment, including the filtration apparatus, introduced only minimal concentrations of the various inorganic chemical species to the samples. With the exception of relatively small levels of toluene in a few instances, the sampling procedure employed contributed only a minor amount of purgeable organic contamination. As well, laboratory procedural blanks (i.e., organic-free deionized water extracted in the usual manner), and field sampling blanks contained insignificant quantities of extractable organic contaminants. Thus, no significant contamination of samples during collection, storage or analysis is apparent.

REPLICATES.

Table 5 lists the sampling points at which replicate samples were taken. Agreement among replicate analyses was generally quite good, as the data in Appendix C demonstrate. The reproducibility for the extractable organics analyses

was judged to be satisfactory in a qualitative sense by the visual comparison of replicate chromatograms.

The accuracy and precision of the analysis of the purgeable organics was further assessed through a set of ten replicate determinations for a standard mixture of nine organics (see Table 6). Coefficients of variation for all nine compounds were found to be acceptably low ($<15\%$). The results from these replicate analyses were compared to their theoretical or "true" values (weight of organic added to standard aqueous mixture solution) by a standard, two-tailed t-test (Zar, 1974). As is shown in Table 6, the mean concentrations of eight of the nine species were not significantly different from their theoretical values ($P>0.5$ for all eight) indicating a good recovery of each from solution during analysis. The sole exception was naphthalene, which exhibited a mean concentration significantly below the calculated value ($P<0.001$). Levels of naphthalene in groundwater samples from the site were usually much lower than those of most of the other purgeable organics, hence this discrepancy was not a major concern. However, quantitation of naphthalene in groundwaters was not reported because of its low recovery upon analysis.

INTER-LABORATORY COMPARISON FOR PURGEABLE ORGANICS.

In addition to determining the accuracy and precision with which the individual purgeable organic compounds were quantified, it is also important to assess the reliability

of the identification of each of the compounds. The use of a gas chromatograph/mass spectrometer (GC/MS) on replicate samples provided an independent confirmation of individual species identities and concentrations.

Table 7 summarizes the results of the UW/MTL inter-laboratory comparison. The interpretation and quantitation of most of the raw GC/MS data was carried out by the University of Waterloo, with the assistance of MTL scientists. MTL performed the evaluation of data from UW12-3 and UW27-6. For the majority of identifications, agreement between the two laboratories is quite good. This implies that a very high proportion (perhaps all) of the individual peaks may be accounted for by the presence of the identified compound. Notable exceptions were encountered in the samples exhibiting the highest concentrations of purgeable organics, especially UW28-2 and UW29 (the seep near UW7). In these instances, the GC/FID concentrations usually exceeded those obtained by GC/MS, due to the presence of high concentrations of co-eluting (i.e., interfering) species. Nevertheless, the data in Table 6 are encouraging because they suggest that the reliability of the purgeable organics identifications was highest where it was most critical - at sampling points where landfill leachate contamination was not clearly indicated by most other parameters.

reports are included in Appendix D along with the reports of Ms. M.G. Foster concerning the results. Most of these results are included in Tables 8 to 10 as well.

Purgeable Organics.

Table 8 summarizes the qualitative and quantitative results from the GC/MS analysis of the purgeable organics fraction. Qualitative concentrations are indicated as very minor (V = < 1% of total peak areas), minor (t = 1 to 10% of total peak areas), and major (m=> 10% of total peak areas). Many significant peaks were not identified so the absence of a compound cannot be inferred from its absence from Table 8. The concentrations of halogenated, purgeable organic compounds are generally quite low - less than 20 ug/l in all cases. The bulk of the purgeable organic fraction consists of hydrocarbons, ethers, ketones and a few sulfur-containing compounds. GC/MS is often unable to distinguish unequivocally among isomers (the same molecular formula but different molecular structures) of aromatic hydrocarbons. Thus, many such compounds are reported as molecular formula (C_7H_{16} , C_9H_{12} for example).

Tetrahydrofuran (THF) was a major component in many of the samples analysed. This compound is a principal ingredient in most PVC glues and, as such, is commonly encountered in samples taken from PVC-glued wells (Boettner et al., 1982). However, three lines of evidence indicate that most of the THF observed was present in the groundwater, rather

Some of the phthalate esters could have originated from the landfill but the presence of phthalates in piezometer and sampling materials (Appendix E) cannot be excluded as the most common source of these compounds.

Next to phthalates, the compounds most frequently encountered in the base/neutral fraction were benzothiazole and derivatives/substituted forms such as mercaptobenzothiazole. Benzothiazole is commonly used in the manufacture of dyes (Kirk-Othmer, 1978) and rubber products (Appendix E). There appears to be no documented natural sources for these heterocyclic species. High concentrations of benzothiazole were found in water allowed to equilibrate with piezometer materials over a period of several weeks (Appendix E). However, two lines of evidence suggest that the well materials are not the sole source of this compound. First, benzothiazole was not detected in all of the sampling points. If the well materials had been the only source of the compound, it would have been found in all of the samples. Secondly, benzothiazole is manufactured by the reaction of N,N-dimethyl aniline with molecular sulfur (Merck, 1983, p. 1108). The presence of all of these compounds in many samples from this site (Tables 9, 10 and Appendix E), suggests that some of the benzothiazole originated as an industrial waste product.

Carboxylic and benzoic acids are common in sampled groundwaters (Table 10). They result from the biological breakdown of organic material in landfill (Robinson and Mar-

is, 1981) but can also be produced biologically in uncontaminated groundwaters. They are in turn broken down to CO_2 and CO_2 plus CH_4 by bacteria in groundwater. This involvement in complex microbiological processes makes an interpretation of organic acid distribution in groundwater very difficult.

Other sulphur-containing heterocyclic compounds could have originated from an industrial source. Stuermer et al. (1979) detected trithiolane, trithiane, tetrathiolane and pentathiepane at relatively high concentrations ($10^1 - 10^2$ ug/l) in groundwater within the burn cavity created by an in situ coal gasification project, but detected none of these compounds in groundwaters taken from undisturbed coal seams away from the gasification site. Their results suggest, but do not prove, an industrial, rather than a natural source for related compounds (trithiane, trithiolane, hexathiepane) found in groundwaters at the Upper Ottawa Street site.

Temporal Variability of Groundwater Quality

The study of piezometer flushing protocol identified considerable short-term temporal variability in the chemistry of sampled groundwater. The routine monitoring of TOC and Cl indicates considerable long-term temporal variation exists as well. Initial variation could reflect contamination by drilling fluid or by water standing in the borehole

prior to piezometer installation. However, much of the observed variation persisted after flushing of such water and must reflect real temporal variation.

Figure 2 indicates some typical patterns of variations in the routine monitoring parameters TOC and Cl. Concentrations are plotted as log values and so these graphical presentations dampen variations. This variability could reflect the complexity of fracture flow paths near the piezometers which causes mixing of chemically-different water masses during pumping or sampling. Certainly the interpretation of the spatial distribution of chemical parameters is more difficult when such temporal variability exists.

Spatial Distribution of Water Quality Parameters

Figures 4 to indicate the spatial distribution of selected water quality parameters in groundwaters near the landfill site. Only the results of the most recent analyses (July - December, 1983) are shown, since these represent the most thorough "snap shot" of the groundwater chemistry at this site. Figure 3 locates the piezometers and seeps sampled.

Waters emanating from sanitary landfills often show elevated temperature reflecting the heat generated by microbial processes in the landfill. The highest groundwater temperatures were recorded on the landfill. However, the sampling technique produced so much variability in tempera-

ture measurements that this thermal anomaly could not be used to trace leachate migration.

pH values shown in Figure 4 of both leachate and groundwaters were consistently near neutral (6-8). Many landfill leachates are more acidic, reflecting the input of organic acids, but the abundance of carbonate minerals in surficial deposits and bedrock at this site encourages the neutralization of any acidic waters.

The electrical conductivity (Figure 5) reflects the total dissolved ionic loading of waters. All groundwaters have high conductivity (>1000 uS) and many exceed normal sea water values of about 50,000 uS. Leachate (UW26-1, 28-2, 29-1, for example) ranges from 13,100 uS to 25,900 uS. Much higher values are found well-removed from the landfill reflecting, in some cases, the high inorganic loading of uncontaminated groundwaters. It appears that shallower groundwaters have, in general, lower conductivity values than groundwaters from the deeper shales and carbonates.

Leachate samples (UW26-1, 28-2, 29-1) have relatively low concentrations of calcium and magnesium ions (Figures 6 and 7) compared to many nearby groundwaters. This suggests that high concentration of these ions is natural, probably due to natural dissolution of Ca^{2+} , Mg-carbonates by groundwaters.

Leachate samples contain considerable sodium, but again much higher Na^{+} values are seen in deeper groundwaters, sometimes well-removed from the landfill site (Figure 8).

Naturally high Na^+ values could reflect leaching of Na-minerals, especially NaCl, from deeper bedrock formations. A similar picture for chloride (Cl^-) is seen in Figure 9. Potassium is elevated in the leachate compared to most groundwaters, although again, high K^+ values appear in deep groundwaters such as at UW20-5, -6 (Figure 10).

Leachate-contaminated samples UW28-2 and 29-1 have very high alkalinity values, probably reflecting the input of inorganic carbon (CO_2 , HCO_3^-) to groundwaters from biodegradation of organics in the landfill site. High alkalinity occurs at UW1-2, UW7-1 and UW22-1, but elsewhere alkalinity falls to about an order of magnitude lower (Figure 11). These lower levels are not unexpected in a carbonate-rich bedrock where alkalinity is generated by dissolution of calcite and dolomite.

Sulphate is usually low in landfill leachates, reflecting the anaerobic microbial environment in landfills which fosters sulphate reduction to H_2S or HS^- . Natural groundwaters could also have undergone sulphate reduction - all but a few shallow groundwaters are devoid of oxygen (Figure 12). Anaerobic sulphate reduction could occur naturally. Almost all groundwaters contain an order of magnitude more SO_4^{2-} than is observed in present landfill leachate (Figure 13.) Noteable exceptions are UW14-3, 9-3, and 17-2. The higher sulphate contents again reflect typical water quality in carbonate-shale-evaporite bedrock.

Fluoride (F^-) and boron (B) are rather high in leachate samples - up to 6.6 mg/l and 120 mg/l respectively in UW28-2 - but are generally less than 1 mg/l and 8 mg/l respectively in groundwaters (Figures 14 and 15). For reference, sea water contains about 1.3 mg/l F^- and 4.5 mg/l B. Both F^- and B are naturally occurring and so could reach high levels in natural brine-like groundwaters as well as in leachate-contaminated waters.

Nitrate (NO_3^-) levels are low throughout this area (except in UW20-5) as would be expected in anaerobic waters (see Figure 16). However, the reduced nitrogen species NH_3 and Kjeldahl nitrogen are extremely high in landfill leachate and high in many groundwaters (see Figures 17 and 18). Some high reduced-N levels probably indicate leachate contamination, but others probably reflect the high level of N-containing dissolved organic matter naturally present in the organic-rich bedrock.

Total organic carbon (TOC) and dissolved (filtered) organic carbon (DOC) measurements show that there is considerable organic matter present in both leachate and some groundwaters (Figures 19 and 20). Such high levels of TOC/DOC cannot be attributed solely to landfill leachate contamination because natural groundwaters could contain high levels as well.

Landfill leachate is typically high in iron, reflecting the enhanced solubility of Fe in reducing, organic-rich

waters. Some of the surrounding groundwaters are even higher in Fe (up to 26 mg/l) as seen in Figure 21. As with most other constituents Fe has natural sources such as pyrite in the shales and carbonates and so its presence, even at high levels, does not prove landfill leachate impact.

Manganese (Mn) levels are usually much lower than iron in leachate and in most groundwaters (Figure 22). Trace metal concentrations are usually low as well Arsenic (As), cadmium (Cd), lead (Pb), selenium (Se) and zinc (Zn) rarely exceed common drinking water quality criteria, even in the leachate. Chromium (Cr) often exceeds the criteria of 0.05 mg/l (Figure 23) with some values up to 1-6 mg/l (UW23-4, UW22-5 and leachate at UW28-2). Naturally high (>0.05 mg/l) levels of Cr are not expected in groundwaters in this area and so most high Cr levels probably originate from man-made sources.

The geochemical control of trace metal mobility via reactions such as sorption and chemical precipitation must be appreciated before metal distributions are evaluated. Figure 34 from Cherry et al. (1984) provides a simplified view of the influence of pH and redox potential (pe or Eh) on selected trace metals. Many metals such as Pb, Cd, and As will be immobilized by precipitation as sulphides in landfill leachates. Cr should also show little mobility at near-neutral pH because of precipitation of Cr_2O_3 . The

occasional high levels of Cr in groundwaters at this site are therefore somewhat surprising and should be confirmed by further analyses. Cation sorption should also greatly retard the migration of most metals, but Cr and As can form anions which are not adsorbed and so may be very mobile in groundwater.

The distribution of trace metals in these groundwaters suggests little impact from landfill leachate except, perhaps, for Cr. This could reflect the geochemical controls of precipitation and sorption and/or it could reflect minimal levels of all but Cr in the landfill material.

Methane (CH_4) is commonly generated in landfill sites but is also present in bedrock accumulations. Figure indicates that CH_4 is common in groundwaters at this site. At 25°C the solubility of methane is about 20 mg/l or 27 cm^3/l . None of the groundwater samples are saturated with this gas. Observed concentrations present no environmental liability. The presence of CH_4 does, however, point to strongly reducing conditions in groundwaters in this area.

Purgeable or volatile aromatic hydrocarbons represent a substantial portion of the identifiable organic matter in the leachate. The distribution of eight such organics was determined (Figures 25 to 32). As discussed in a later section, all but chlorobenzene probably occur naturally in the bedrock, so their presence is not proof of landfill leachate contamination. Their distributions are irregular, with highest concentrations of tens to hundreds of ug/l or ppb.

The distribution of benzene (Figure 25) is typical of these compounds. Leachate contains from 9 to 55 ug/l. Although no drinking water criteria for benzene exists in Ontario, the New York State Standard for Groundwater used for drinking water supply sets 10 ug/l as the maximum limit for benzene. Most groundwaters contain less than 10 ug/l but occasionally higher concentrations (45 ug/l at UW23-4 and 16 ug/l at UW14-3) are found removed from the landfill.

The distribution of tritium (^3H) in selected groundwaters is shown in Figure 33. Tritium is naturally present in water molecules in the atmosphere. After about 1953, natural tritium levels of less than 10 T.U. (tritium units) were replaced by increased levels because of atmospheric nuclear testing and other nuclear activities. Levels of 10^3 to 10^4 T.U. were found in precipitation from 1954 to 1966; the levels steadily decreased to 30-100 T.U. at present. Tritium has a half-life of about 12.3 years. Groundwater recharged before 1953, then, presently contains less than 2 T.U. while waters recharged after 1953 will generally have 10-200 T.U. at present. Thus the tritium level in groundwaters "labels" water recharged pre- and post-1953.

Most of the landfilling at this site occurred post-1953 with the known disposal of industrial waste occurring later. Thus tritium is a useful tool to distinguish the presence of pre-landfilling (pre-1953) from post-landfilling (post-1953) groundwater. Groundwater containing less than 2 T.U. prob-

ably has at most, only a very minor component of post-1953 groundwater. Such samples should, therefore, be essentially free of landfill contamination.

The distribution of tritium is also irregular, reflecting the complex hydrogeology at this site. At many piezometer locations, deeper groundwaters appear to be younger than some shallower groundwater. This indicates more recently-recharged groundwaters can move to considerable depth where fractures permit.

The Impact of Landfill-Derived Leachate on Groundwater Quality

Background Groundwater Quality.

The impact of leachate contaminating groundwaters near the Upper Ottawa Street landfill site can only be assessed with reference to alterations of the natural or background water quality. The hydrogeologic complexity at this site makes it unlikely that uncontaminated groundwater quality is consistent. At least two end-member background water types can be recognized. One (type 1) is recently-recharged waters which would contain measureable tritium levels and would be rather dilute reflecting the short time for interaction of the recharging rainfall or surface waters with the soil/rock. This water could contain contaminants from sources other than the landfill. The other end-member, (type 2) would have been in the bedrock for a very long time

and so contain high concentrations of many dissolved species, but no measurable tritium. This water would be free of landfill contaminants but would likely be of very poor quality.

Possible examples of type 1 and type 2 groundwaters are listed in Tables 11 and 12 along with a landfill leachate spring for comparison. Type 1 groundwaters almost always contain lower levels of all parameters except Ca and SO_4 when compared to the leachate seep. Note, however, the presence of most purgeable organic parameters and some trace metals in type 1 groundwaters. Type 2 groundwaters, on the other hand, almost always contain higher levels of inorganic parameters (except for alkalinity, Kjeldahl N, DOC/TOC, Fe and Mn) than does the leachate spring. Purgeable aromatics are lower in type 2 background groundwaters, except for toluene perhaps, than in the leachate seep and the xenobiotic chlorobenzene is absent. It is clear that leachate-contaminated groundwater cannot be distinguished from leachate-free groundwater based on the concentrations of one or two species.

LEACHATE CHEMISTRY

Present-day landfill leachate chemistry is summarized in Table 13. This range of concentrations may not be completely representative, i.e. it may be greater than shown.

Since most landfilling was undertaken after 1953 when elevated tritium levels appeared in infiltration, leachate would still have detectable tritium today. Lack of detectable tritium could reflect dilution of leachate by pre-1953, background groundwater or the presence of pre-1953 background groundwater without leachate contamination. Recently-recharged, uncontaminated groundwater would contain tritium and so tritium alone is not a reliable indication of contamination from the landfill.

Unfortunately, the same can be said for all the inorganic parameters determined - a natural source of each is possible. The presence of man-made xenobiotics such as halogenated organics would be an excellent indication of contamination. However, the contamination source could not be identified reliably as the landfill, since other potential sources are present in all urban areas. The initial studies of leachate, groundwaters and surface waters at this site found the organic contaminants to be mainly non-halogenated hydrocarbons. Many, if not most, of these organic compounds can also be present in uncontaminated groundwaters which

have leached these hydrocarbons from the naturally-occurring, bituminous, hydrocarbon-rich, organic matter ubiquitous in the bedrock.

Recognition of Leachate-Contamination in Groundwaters

A subset of the large number of organic molecules possibly present in landfill-leachate was selected for detailed sampling and analyses in order to define the extent of landfill-leachate contamination of groundwater at the Upper Ottawa site. We focused on volatile (purgeable) aromatic organic compounds because they are a major component of landfill leachate, they are relatively soluble and hence mobile, and they are rather recalcitrant or persistent under anaerobic conditions expected in leachate plumes (Wilson and McNabb, 1983).

Purgeable aromatics are naturally-occurring as well, being common constituents of natural crude oil and bitumen. These low molecular weight aromatics are among the most water soluble compounds in these organic materials and so are expected in associated groundwaters. Aromatic hydrocarbons form a significant portion of naturally-occurring sedimentary organic matter. For example, Tissot and Welte (1978, p. 351) state that benzene can form up to 1% of crude oil while toluene and xylenes can make up 1% to 3% of the crude oil. Naphthalene and other alkyl benzenes are common. Stuermer et al. (1979) identified toluene, ethylbenzene,

xylenes, propylbenzene, 1,2,3-trimethylbenzene, naphthalene and many other aromatics in the water-soluble fraction of Prudoe Bay crude oil.

Similar concentrations of aromatic hydrocarbons can be expected in the bitumen (solvent-extractable) fraction of disseminated organic matter which is geochemically similar to crude oil. This is illustrated by the analysis of an Ontario shale of the Devonian Kettle Point Formation performed by the Chemistry Laboratory of the Technical Support Section of the Ontario Ministry of the Environment. Major volatile hydrocarbon components identified included benzene, toluene, ethylbenzene, and xylenes. Water left in contact with organic rich shales of the Devonian Kettle Point Formation was analysed in our laboratory and the presence of these aromatics, especially toluene, was confirmed. Clearly then, groundwater contacting natural sedimentary organic material can contain these purgeable aromatic hydrocarbons.

Of course, purgeable aromatic hydrocarbons are also common in many refined products which could have been disposed in the landfill or nearby. Zurcher and Thuer (1978) identified many of these aromatics in the water-soluble fraction of fuel oils and gasoline including a spilled oil recovered from the water table of a contaminated aquifer. Stuermer et al. (1979) report similar findings in groundwaters contaminated by in situ coal gasification.

Therefore, the presence of aromatic hydrocarbons in groundwaters near the Upper Ottawa Street landfill site cannot be taken as proof of contamination by landfill leachate. Chlorinated aromatics, such as chlorobenzene, are not found in natural organic material and so chlorinated aromatics may be a much more reliable indicator of contamination but the source may not be the landfill site. As with essentially all other geochemical data, the degree of groundwater contamination due to landfill leachate can only be interpreted.

A comparison of background groundwater chemistries (Table 11 and 12) with leachate chemistry (Table 13) indicates that:

1. leachate is intermediate in terms of total inorganic loading (conductivity)
2. Mg dominates Ca in leachate, Ca dominates Mg in background waters.
3. Leachate has the highest alkalinity
4. All waters are low in NO_3 , and leachate is highest in reduced nitrogen (Kjeldahl N)
5. Many waters contain high levels of DOC/TOC - leachate is again highest.
6. Background and leachate generally have low concentrations of metals with the exception of Fe which is highest in the leachate.
7. All waters contain volatile aromatics, but only leachate has concentrations above a few ppb.

8. Chlorobenzene appears in leachate samples only, but not in all leachate samples.

Figure 35 shows the relationship between Mg^{++} and K^+ in groundwaters. Landfill leachate, type 1 and type 2 background groundwaters from Tables 11, 12 and 13 are highlighted. Leachate has the highest K/Mg ratio. Groundwaters with K/Mg less than about 1/4 could be interpreted as simple mixtures of type 1 and type 2 uncontaminated groundwaters. Those with K/Mg greater than about 1/2 plot near leachate samples and could be interpreted as containing a significant leachate component. Proximity to the leachate samples in Figure 35 could indicate relative level of leachate contamination. This provides another parameter to identify leachate-impacted groundwater.

Plots of other pairs of inorganic parameters were made to help identify a leachate component. A plot of Mg/Ca ratio versus K was of little use but plots of Cl versus SO_4 and Ca versus alkalinity seemed useful. These are shown in Figures 36 and 37. The degree of leachate impact is indicated by proximity to the leachate samples and departure from the trend between type 1 and type 2 uncontaminated groundwaters.

The purgeable organic compounds in groundwaters were also incorporated into this leachate-impact-identification process. Where the sums of the aromatic hydrocarbons exceed 20 ppb, leachate impact was considered at least "weak"; when

the sum exceeded 100 ppb the leachate impact was "strong". Where chlorobenzene was detected, a "weak" impact was assumed except where chlorobenzene content exceeded 1 ppb at which point a "strong" impact was assumed.

Although far less reliable and comprehensive, the GC/MS analyses of volatile organics, was considered as well. The presence of significant chlorinated compounds (more than a trace of 1 or 2 compounds or a trace of more than 2 compounds) indicated weak leachate impact; > 10 ppb total chlorinated organics rated a strong impact designation. Similarly, the presence of the common O-containing volatile solvents - tetrahydrofuran (THF), 2-butanone (methyl ethyl ketone or MEK) and acetone at trace levels (weak impact) and significant or major levels (strong impact) was considered.

Criteria were also defined in the base/neutral and acid extractable fractions analysed by GC/MS. The presence of traces (weak impact) or of more significant levels (strong impact) of benzothiazole and polychlorinated biphenyls, ethers, halogenated-organic phthalates, carboxylic acids and aromatic acids provided more criteria.

This resulted in three inorganic tests, four tests using volatile or purgeable organics and seven tests using extractable organic data - fourteen tests in all to apply to the problem of identifying leachate impact in groundwaters. Each groundwater was then rated with strong, weak and no leachate impact given scores of 1.0, 0.5 and 0.0 respective-

ly and the final rating expressed as a % of the maximum possible score (1.0 for each test available for that sample). Results are shown in Table 14 along with the number of tests applied to each sample. Although this system is still somewhat arbitrary and selective, it at least considers a wide range of parameters for the task of recognizing landfill-leachate impacted groundwaters.

Interestingly, the type 1 and type 2 background groundwaters shown in Tables 11 and 12 scored 0%, 14%, 0%, 5% and 0%. On the other hand, leachate-contaminated groundwaters 28-2 and 29-1 (Table 13) scored 54% and 68% respectively. A score greater than 14% might be accepted as evidence of significant contamination. Of the 95 sampling points, 45 or 47% scored greater than 14%.

Where no tritium was found, the sample could not have a significant landfill leachate component. UW7-3 was removed from the contaminated list for this reason. The major contamination indicators for UW7-3 had been the high levels of benzothiazole, carboxylic acids and 2-hydroxybenzoic acid and low levels of phthalates. These types of organics are common in the piezometer material (Appendix E). Other groundwaters were also eliminated from the contaminated category, mainly because only a very weak indication of contamination, which may easily have originated from the piezometer material, was found. These typically had only two tests available. Groundwaters from UW18-3, 21-3, and 27-4 showed

organic contamination that could be from piezometer material and type 2 groundwaters and so were also eliminated. Many of the apparent indications of landfill leachate contamination in these piezometers had also been noted in UW7-3 which was not considered to be impacted by the landfill leachate.

The seep at Albion Falls (UW31-1) is listed as impacted by the landfill with a score of 22%. Indicators of contamination were benzothiazole, phthalates and carboxylic acids with only the benzothiazole yielding a strong indication of contamination. A piezometer was not used for sampling this seep so the benzothiazole cannot be attributed to such material. So, we have a groundwater sample which shows one strong indication of leachate impact, two weak indications and six tests which did not indicate leachate impact. The hydrogeological setting of this seep more easily supports Red Hill Creek as the source of contaminants, if they are, in fact, landfill-derived, than the landfill site directly. Also, phthalates have been identified in Red Hill Creek (report by Mann Testing to UOSLSS, Sept. 22, 1983).

Similarly, the list of contaminated groundwaters must be examined in their hydrogeological context. That is, does a likely hydrogeologic link exist between piezometer points and the landfill? The answer is likely to be equivocal, given the uncertainties of groundwater flow paths and velocity in the fractured rock at this site.

Consider groundwater from UW20-5. This deep sample contains tritium indicating a significant component of recently-recharged water is present. The contamination "score" of 32% on 11 tests points to significant contamination. However, there does not appear to be a continuous hydraulic gradient from the landfill to UW20-5. There exists a horizontal gradient from the landfill to UW9 and UW10 but then the gradient reverses, suggesting flow from UW20 towards UW9 and UW10. The most serious contaminants in UW20-5 are benzothiazole and carboxylic acids for which other non-landfill sources, both natural and piezometer-derived, exist. UW20-5 is therefore considered not to be impacted by landfill-leachate.

The Extent and Severity of Landfill-Leachate Contamination

The extent of groundwater contamination is indicated in Figure 38. Away from the immediate area of the landfill, minimal impact upon the already poor background groundwater quality is indicated. The distribution of apparent groundwater contamination is not simple. For example, near the landfill it was anticipated that shallow groundwaters in the relatively permeable carbonate bedrock would show significant leachate impact. At some piezometers this is apparent - UW7, UW17, UW22 for example. At other piezometers contamination only at depth is indicated - UW21 and UW24. It seems unlikely that shallow points at all these latter pie-

zometers have escaped contamination; rather the contamination is difficult to recognize given the limited analyses and the poor quality of the background groundwater.

Another area of contamination is well to the south of the landfill site in the deeper groundwaters at UW14 and UW21. The hydrogeological connection of these groundwaters to the landfill is unlikely. The contamination may only be apparent and due to poor quality background water, or it may not originate at the landfill.

In fact, "strong" evidence of landfill impact can be seen only in piezometers UW7-1, 17-1, 21-5, 22-1, -3, -5, 26-1, 27-1, 28-2, and 29-1. Weaker evidence of leachate impact is seen for UW1-2, 6-1, 7-2, -4, 9-3, 12-3, 14-3, -4, 17-2, -6, 19-4, -5, 13-5 and perhaps also in 23-4, 24-4 and 27-6.

How severe is the groundwater contamination? Table 15 compares the quality of various groundwaters at this site with some drinking water quality standards. Unfortunately, standards for most organics have not yet been established. Included in Table 15 are a landfill leachate (UW29-1), a type I and a type II background groundwater and some impacted or possibly impacted groundwaters. Leachate waters exceed the water quality criteria for many inorganic parameters and for benzene and PCBs. Background groundwater also exceeds the criteria for many inorganics but not for organic parameters. Other, possibly impacted groundwaters, similar-

ly exceed the water quality criteria for many inorganics but not for organics. This comparison suggests that the impact for landfill leachate upon groundwaters is not serious, in that the number of chemical parameters exceeding the water quality criteria is only occasionally higher than for type II background (uncontaminated) groundwater. This interpretation must be tempered by the observation that many organic compounds have been found, in low concentration, in groundwaters at this site but water quality criteria have not been established for them.

The flux of contaminants entering the environment via groundwater can be estimated by combining data on water flux or leachate generation (p. 70) and contaminant concentration in that leachate. Estimates of water flux emanating from the landfill site range from 2.5 to 21 l/s. Contaminant concentrations vary widely over the landfill (see data for UW26, UW28, UW29). Fluxes selected for contaminants are presented in Table 16 as ranges using the above data. These flux estimates are very rough but they do indicate the possible significance of a groundwater pathway. Even a few mg/l of contaminants in leachate results in kilogram quantities emanating via groundwaters each year.

Much higher concentrations of contaminants, especially organics, was anticipated from the history of landfilling at this site. The generally-low concentration of identifiable organics in associated groundwaters could reflect:

1. limited input either because of low organic content of wastes or effectiveness of burning of organic wastes
2. limited leaching by infiltrating due to low water solubility of organics or due to low infiltration flux relative to flux of natural groundwater
3. biodegradation and chemical transformation of organic contaminants in the landfill and along groundwater flow paths
4. "sinking" of a dense organic phase beneath the landfill with subsequent dissolution of organics into groundwater but at a depth below the installed piezometers

Point one, above, is certainly possible. The second factor mentioned above could be true for many compounds especially the oil-derived/petroleum compounds which seem to dominate. However, many of the typical petroleum compounds (alcohols, ethers, aromatic hydrocarbons such as benzene and toluene, and some aliphatic hydrocarbons), have water solubilities of 50 to 100,000 mg/l - many orders of magnitude higher than the observed concentrations in groundwaters. Could these water-soluble organics have been so rapidly leached and rapidly transported that they are beyond the piezometer network already? This is unlikely given the timing of disposal (within the last 25 years), the hydrogeological situation, and the tendency of many of these organics to be retarded by adsorption onto the solid materials. On

the other hand, many of these compounds are sufficiently mobile to have migrated into the groundwater monitoring network rather than remaining in the landfill.

The transformation, either chemically or microbiologically, of many organics likely to have entered the landfill is very possible (Wilson and McNabb, 1983; Kobayashi and Rittmann, 1982). The anaerobic landfill environment may have been very conducive to the degradation of chlorinated, one- and two- carbon organic contaminants such as 1,1,1-Trichloroethane, Trichloroethylene and Tetrachloroethane which are common industrial solvents and degreasers and which are found in surprisingly low concentrations in the landfill, in the leachate and in the nearby groundwaters. On the other hand, most aliphatic and aromatic hydrocarbons persist in such an environment and their low concentrations were unlikely to have resulted from degradation.

Where sufficient quantities of dense, usually halogenated organic liquids are disposed onto land, they may form an organic phase, immiscible in groundwater, which can migrate downward as a discrete phase due to its higher density compared to water. This is an unlikely scenario at this site. The residue of dense halogenated organics does not seem to exist in or immediately beneath the landfill. Such halogenated compounds are never found above trace levels in any groundwaters or leachate suggesting that they were not present in amounts sufficient to establish a dense organic liquid phase.

CONCLUSIONS

The leachate emanating from the Upper Ottawa Street landfill has resulted in only a slight deterioration in currently-defined groundwater quality parameters in the study area. This, in part, reflects the very poor quality of the natural bedrock groundwater. The most serious contamination is limited to within 100 m of the landfill site and is very irregularly distributed. Some contamination is noted to the east of the site - the general direction of lateral groundwater flow. Some apparent contamination of groundwaters up to 500 m south of the site is more difficult hydrogeologically to trace to the landfill site. Even though contaminant concentrations are generally low even in leachate, a significant annual flux of contaminants emanates from this site via groundwater flow. It does appear, however, that dilution and dispersion has greatly reduced the contaminant concentrations in groundwaters near the landfill site. Leachate-contaminated groundwaters do not appear to be migrating towards present areas of housing in the vicinity of the landfill site. The general release of contaminants via groundwater poses a long-term problem, especially since the identity and toxicity of the complex spectrum of organic compounds present in landfill leachate and in natural groundwaters are not well characterized.

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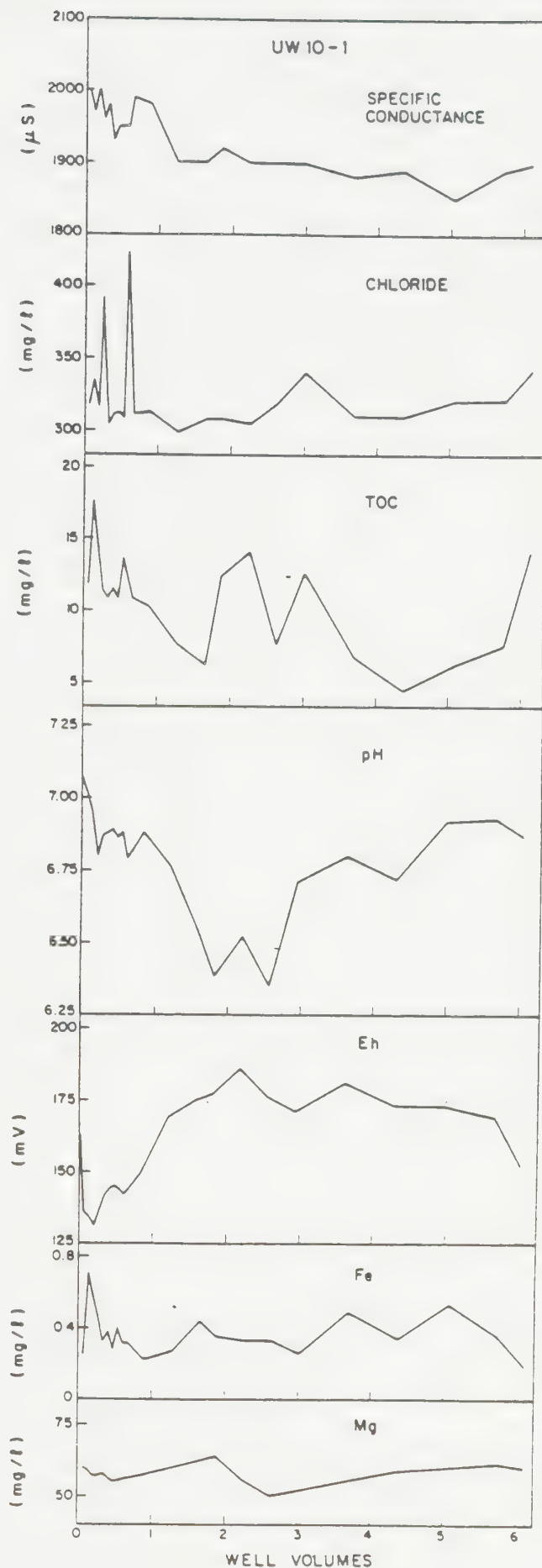
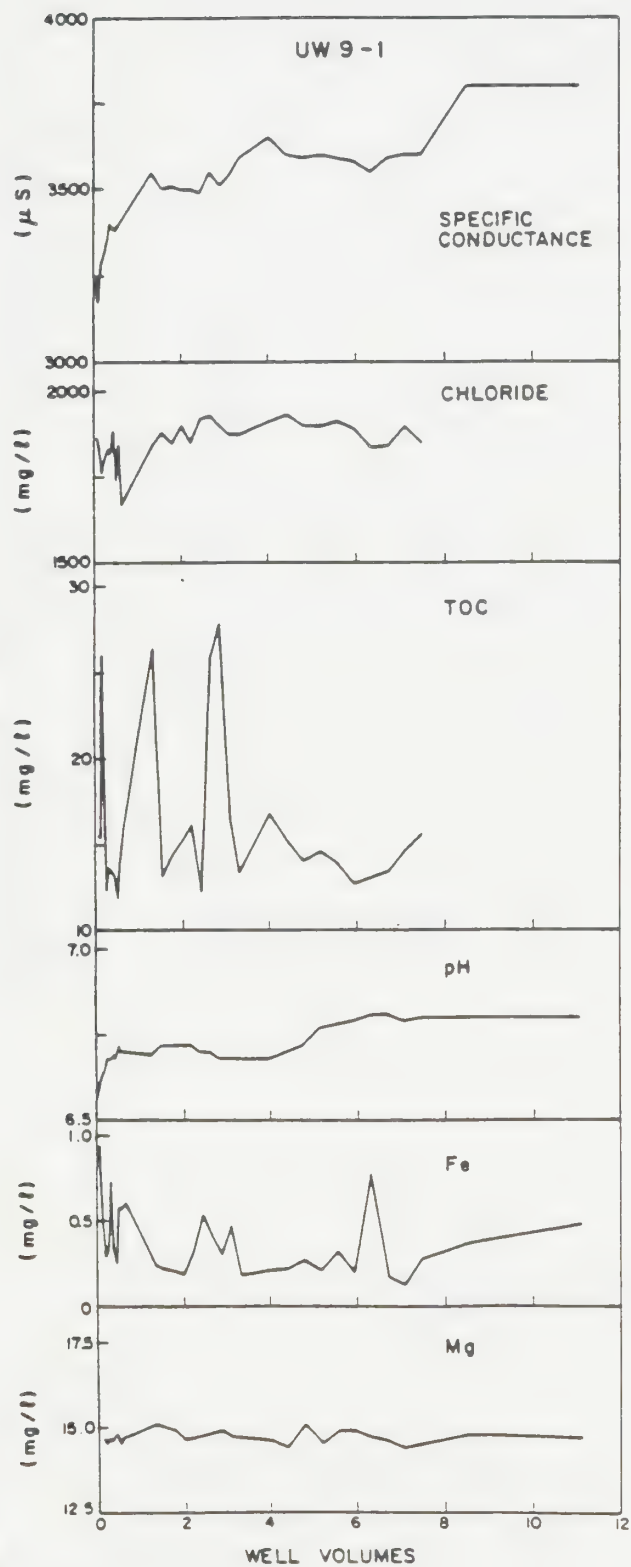
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Figure 1. The variation of selected chemical parameters during flushing of UW 9-1 and UW 10-1.



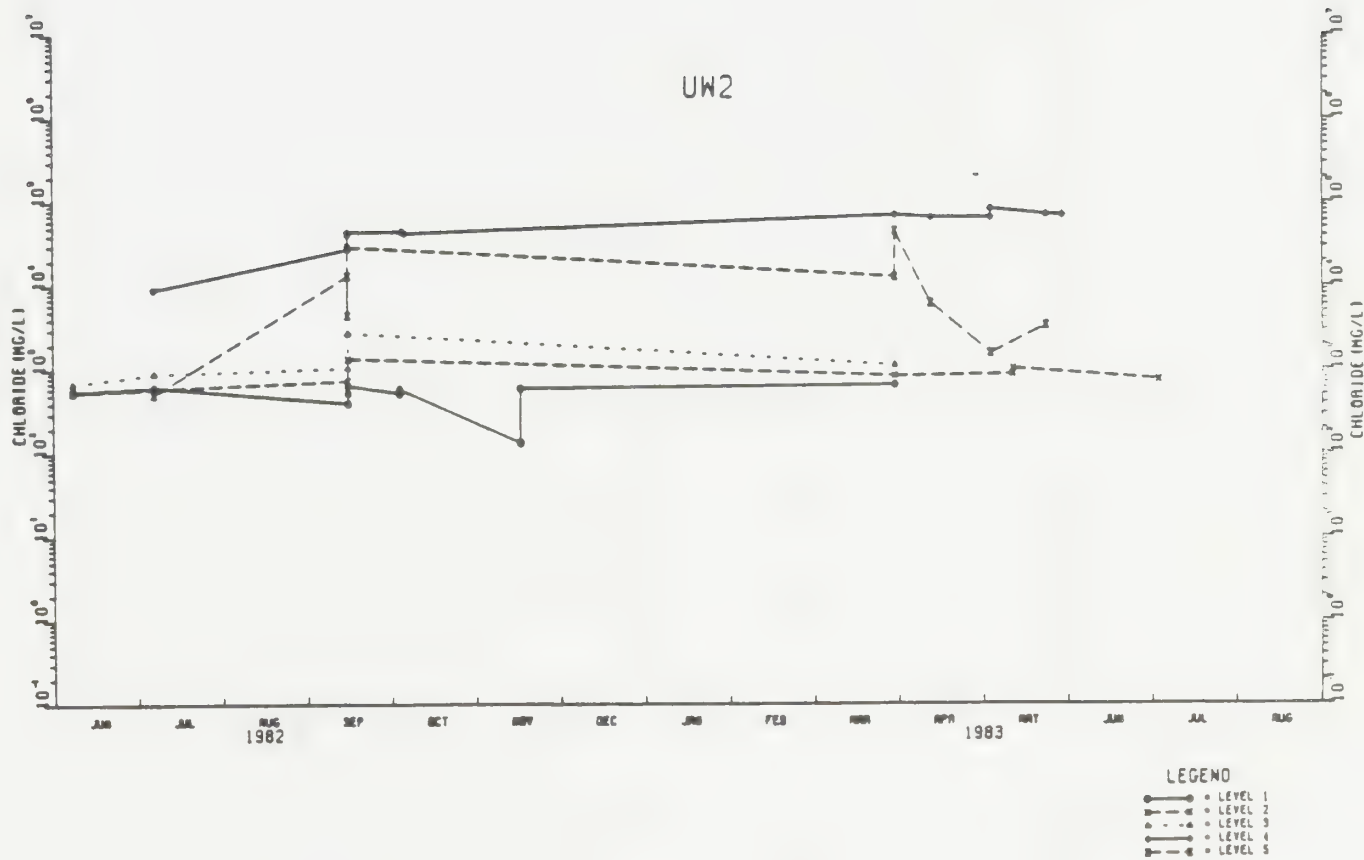


Figure 2a. Temporal variation in the chloride concentration at the depth-specific sampling points in piezometer UW2.



Figure 2b. Temporal variation in the chloride concentration at the depth-specific sampling points in piezometer UW6.

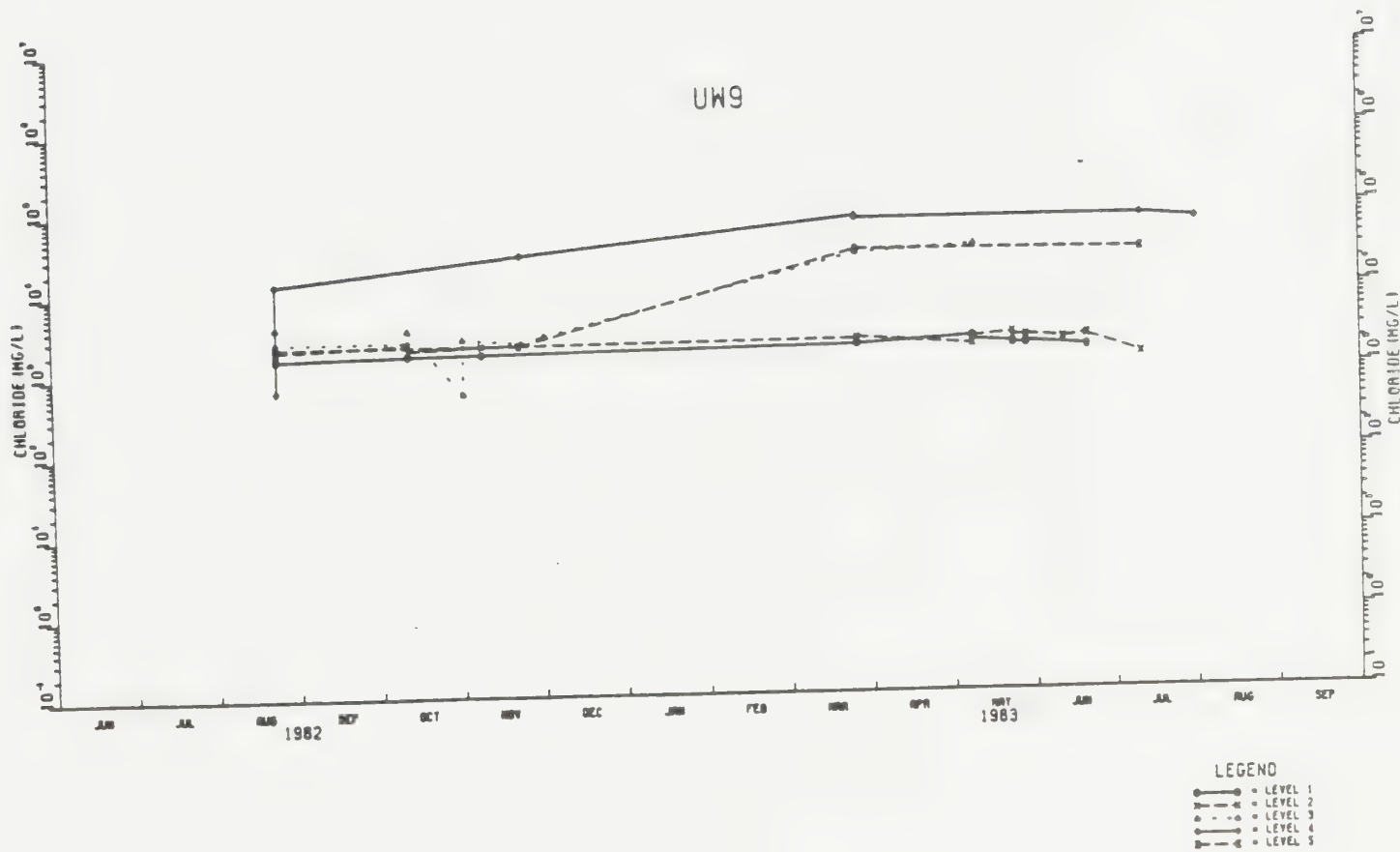


Figure 2c. Temporal variation in the chloride concentration at the depth-specific sampling points in piezometer UW9.

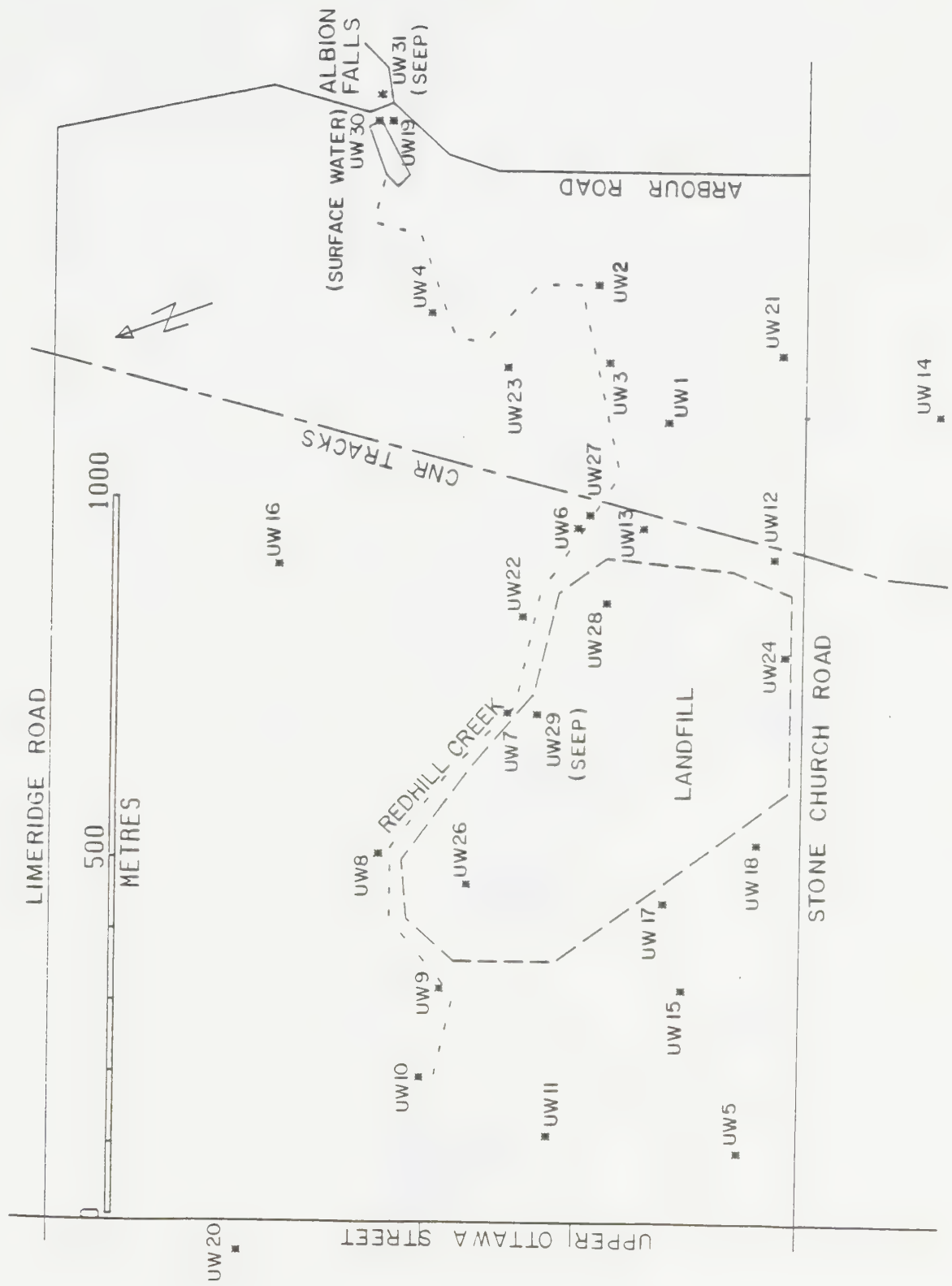


Figure 3. Location of sampling points and piezometers.

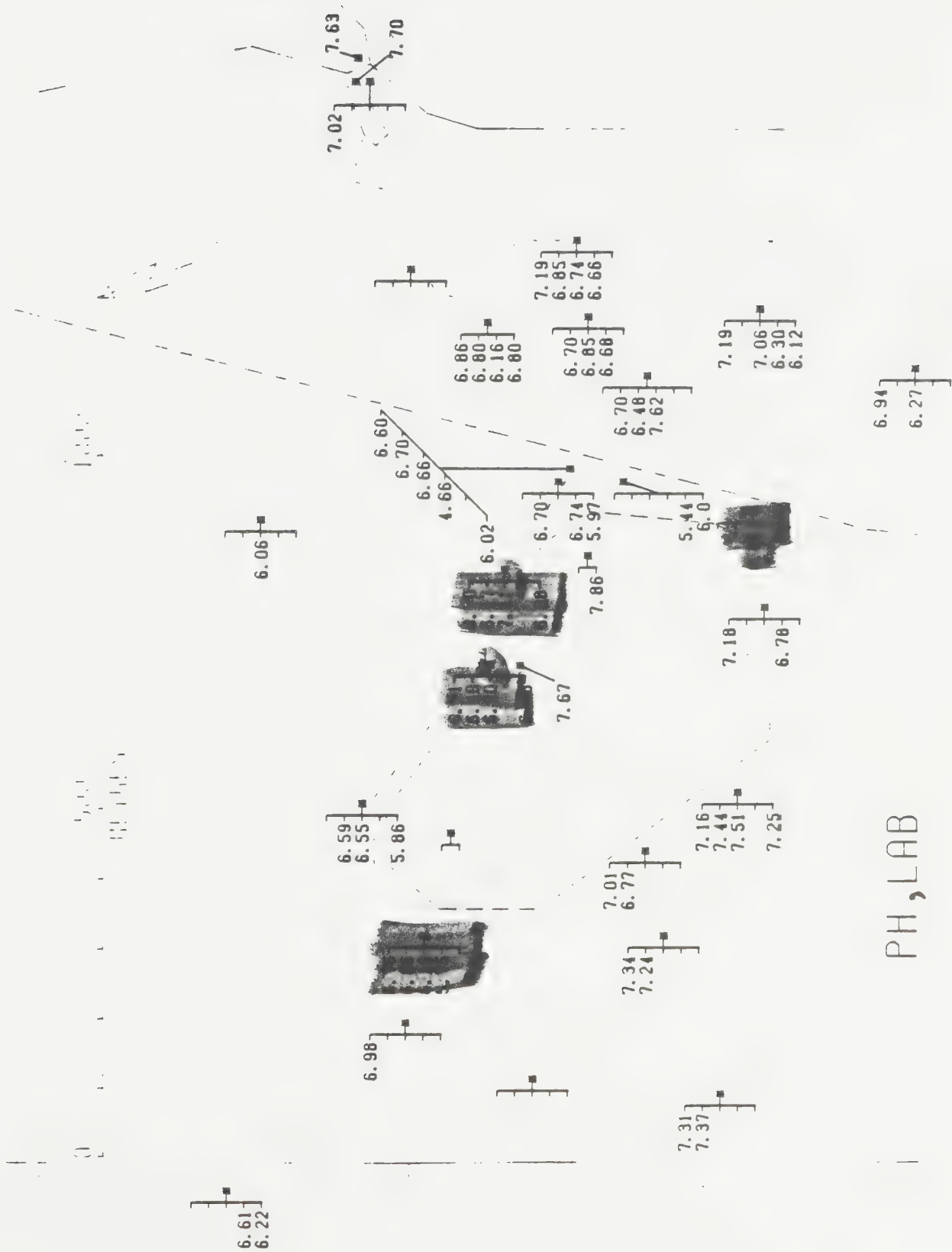


Figure 4.

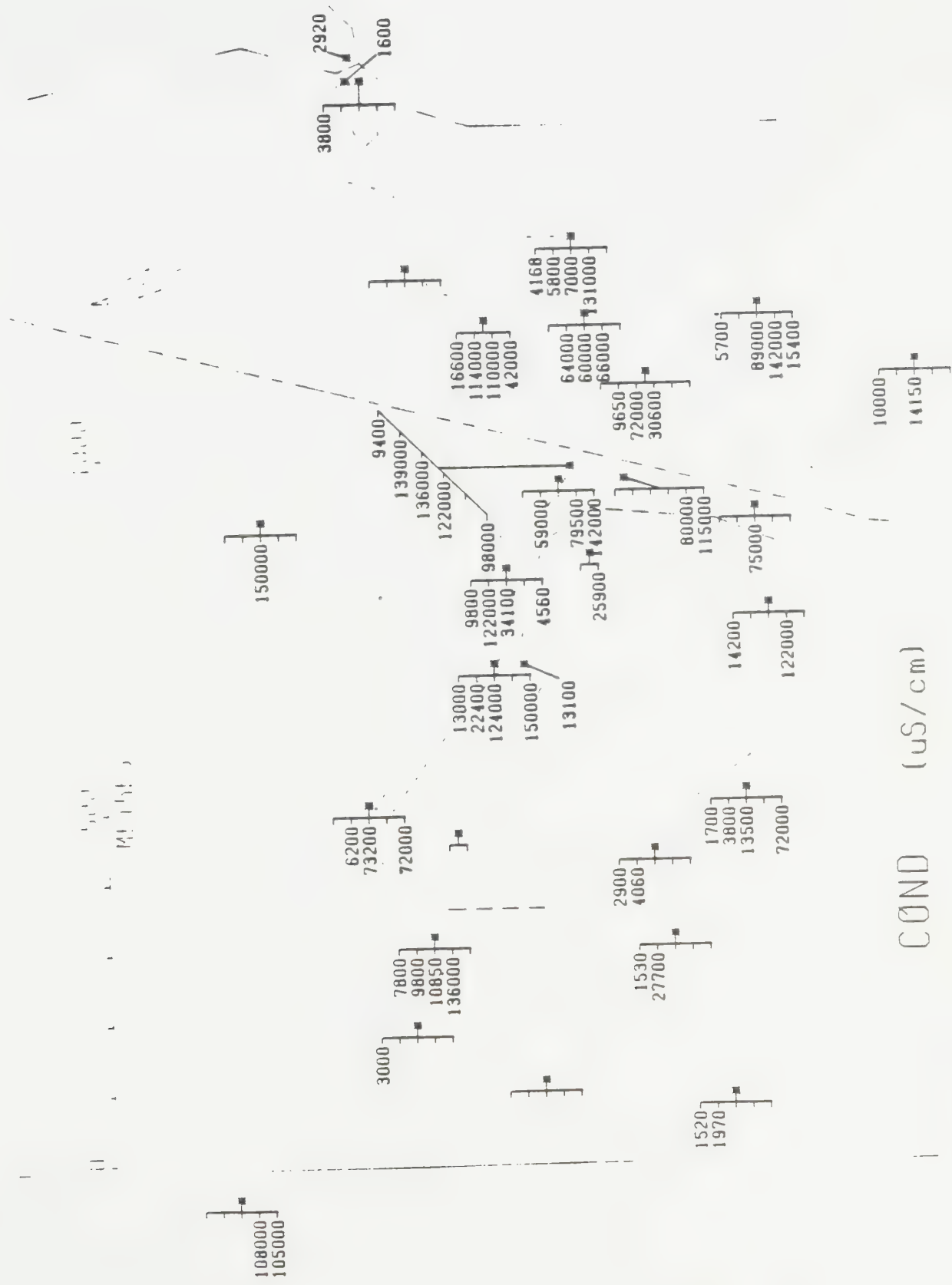


Figure 5.

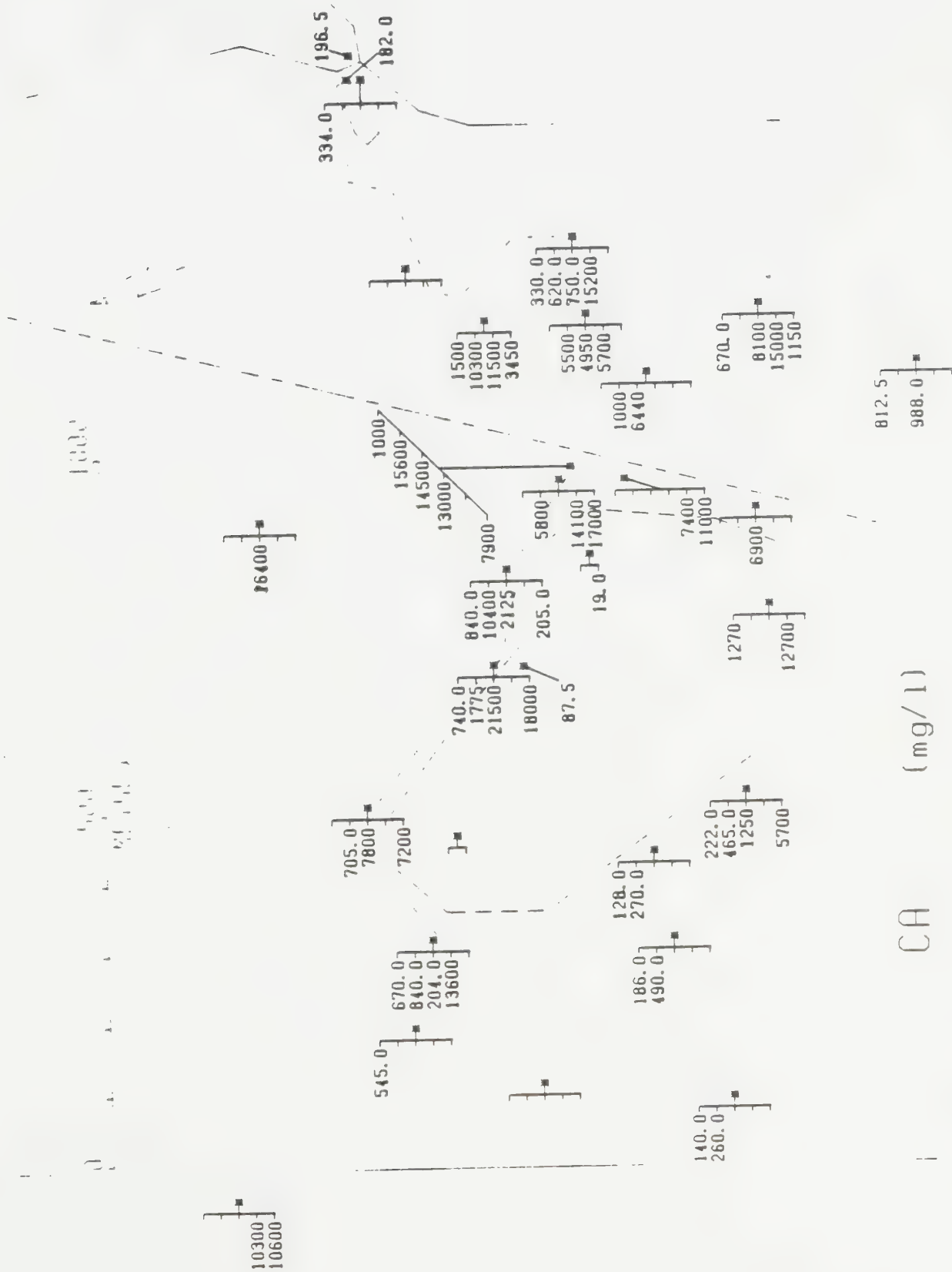


Figure 6.

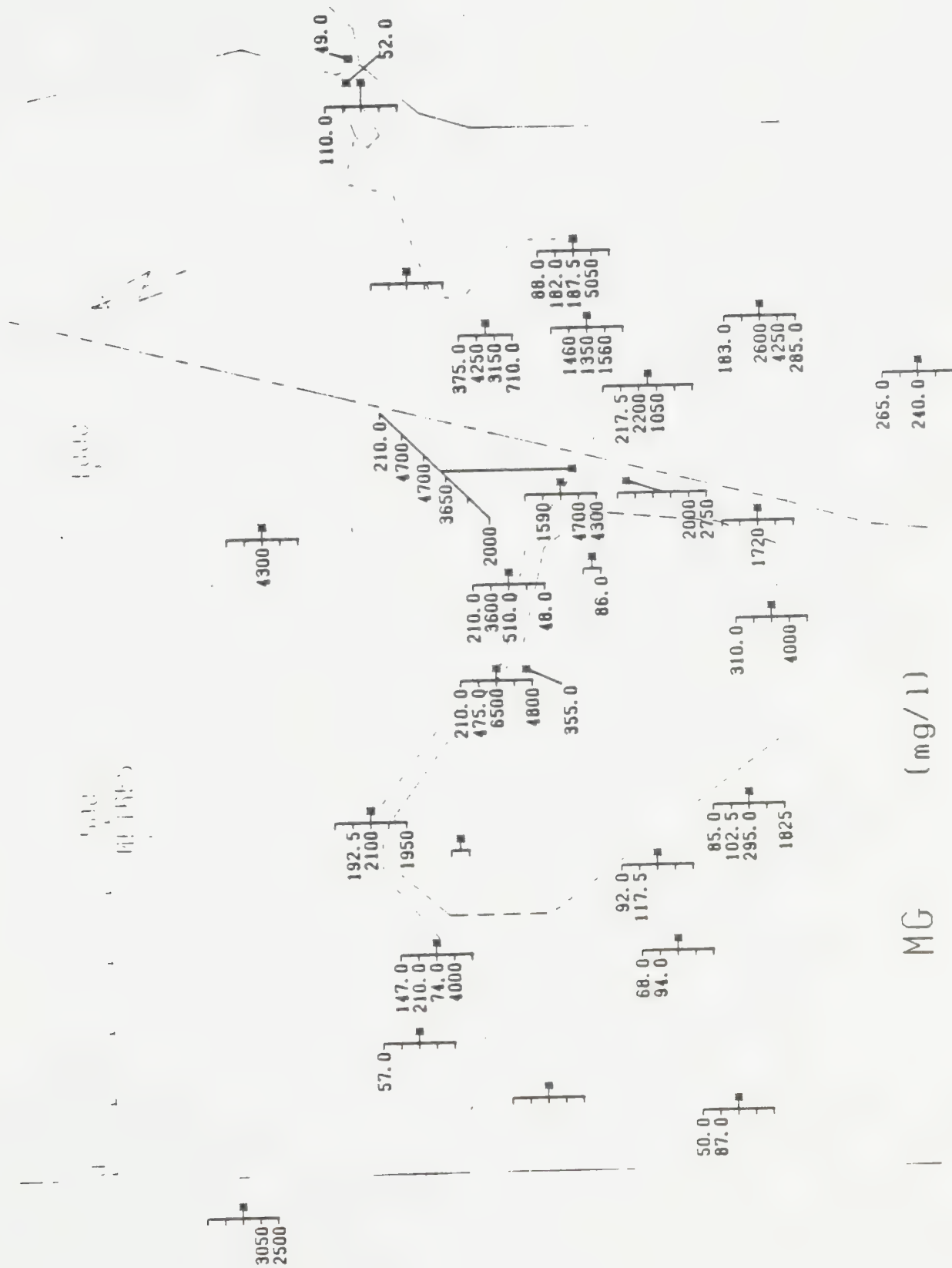
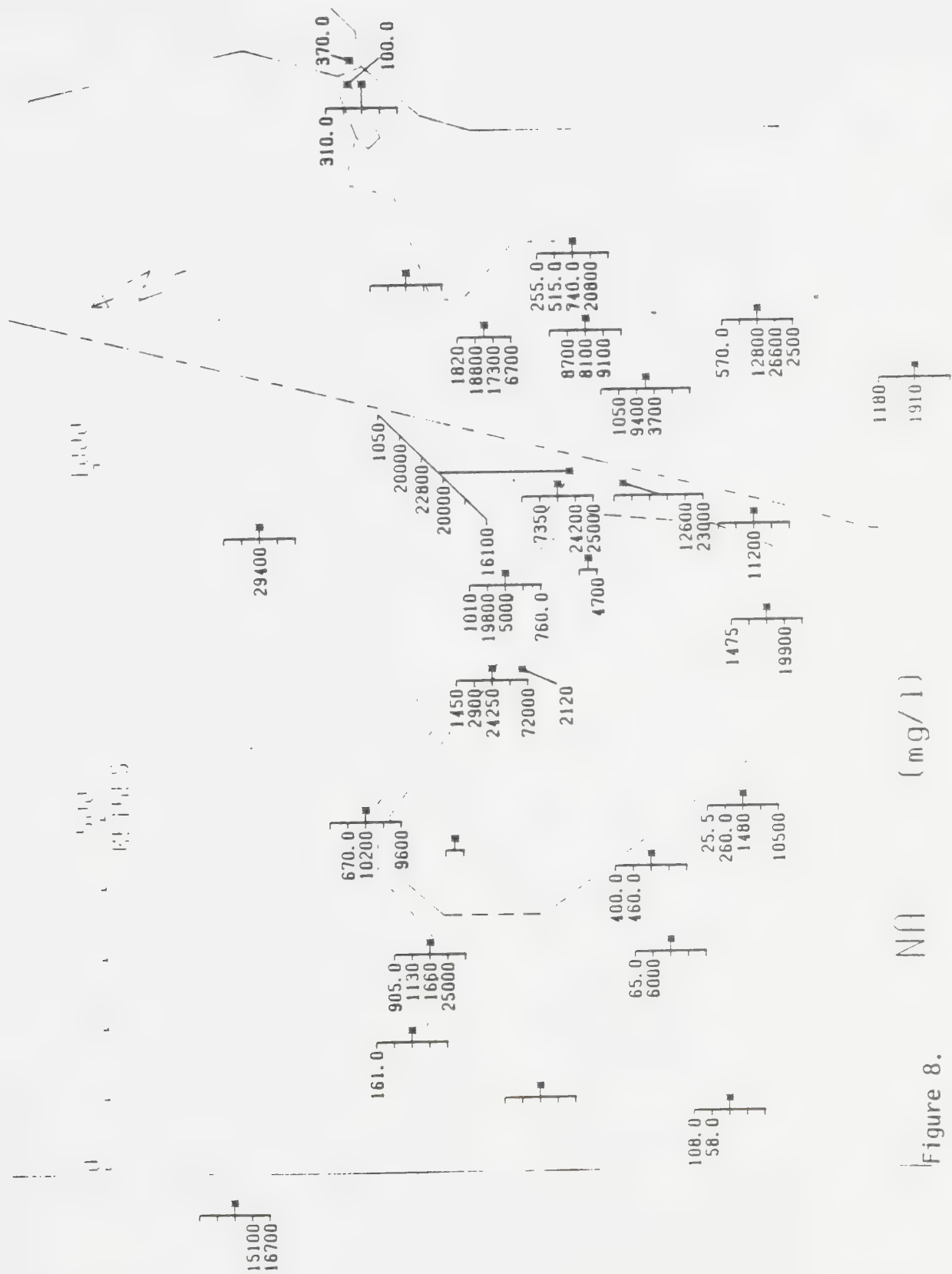


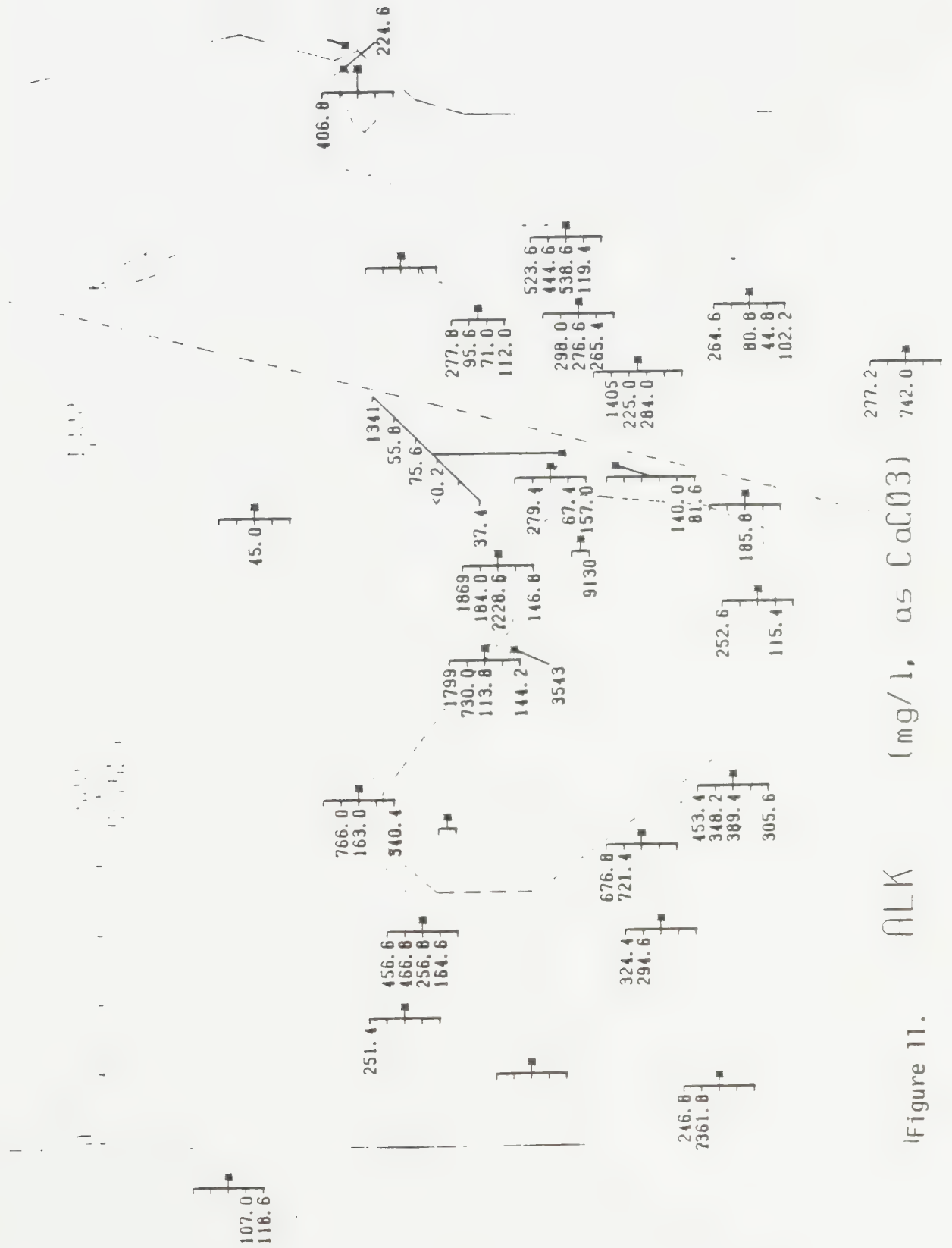
Figure 7.

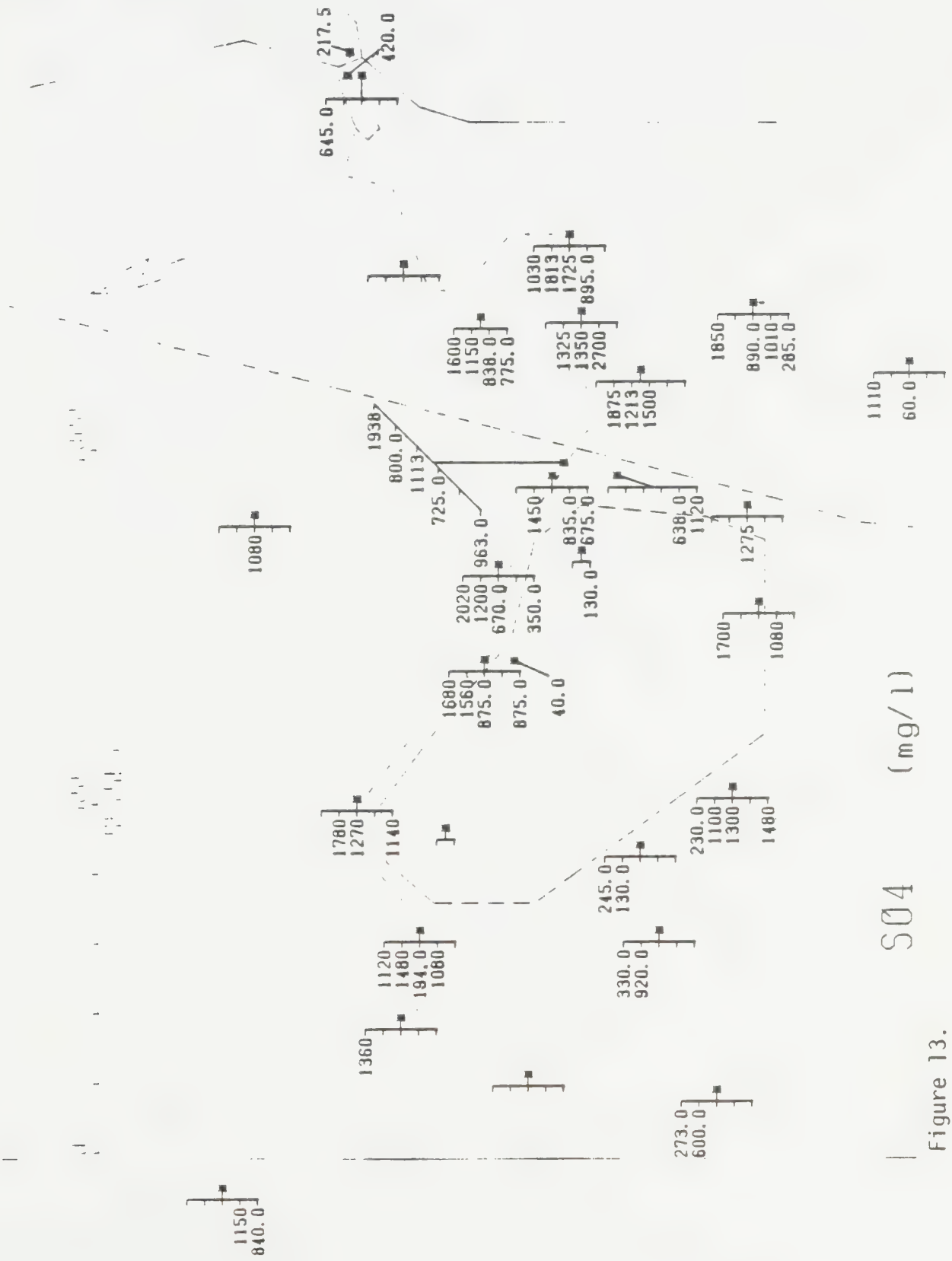


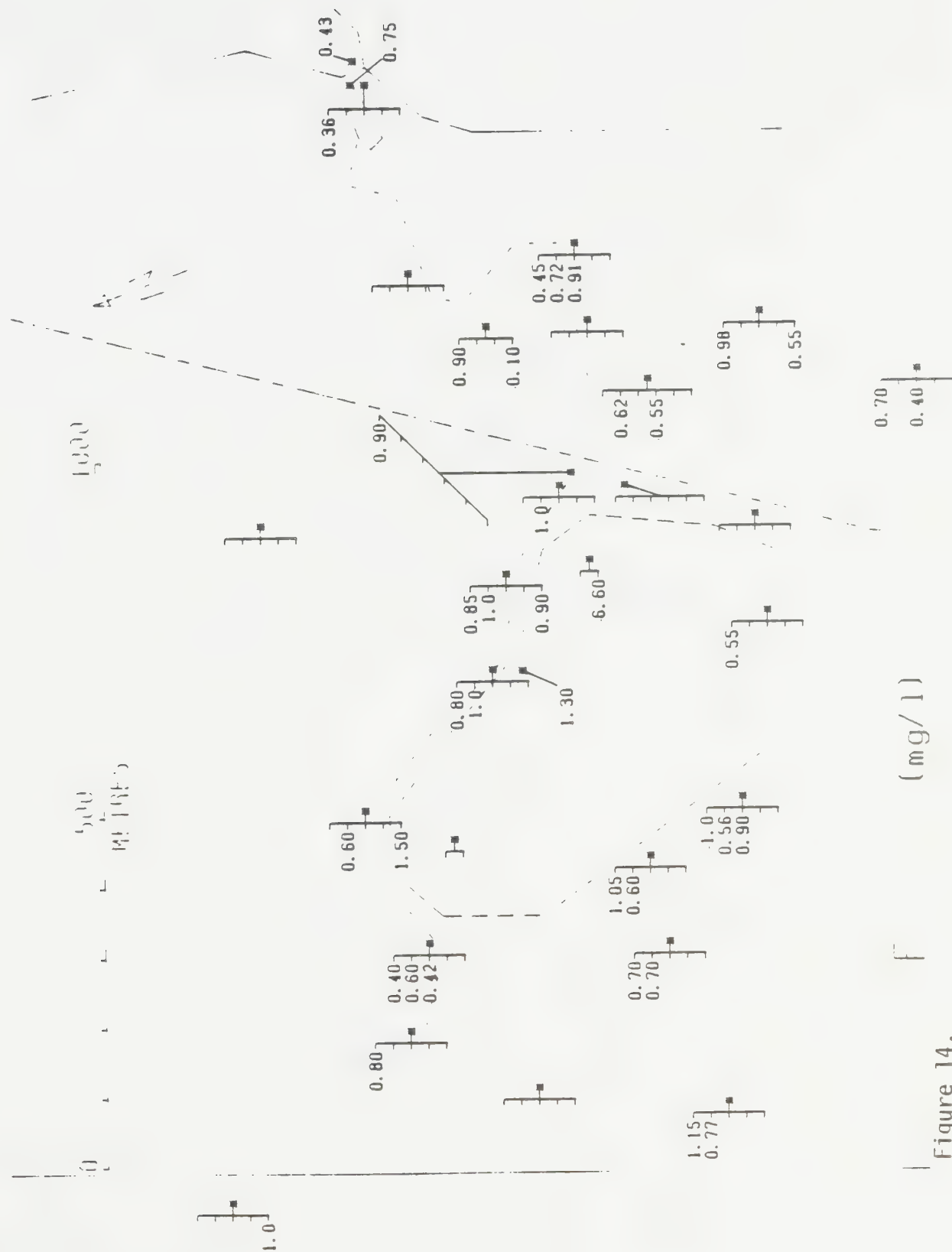
(mg/l)

NH

Figure 8.







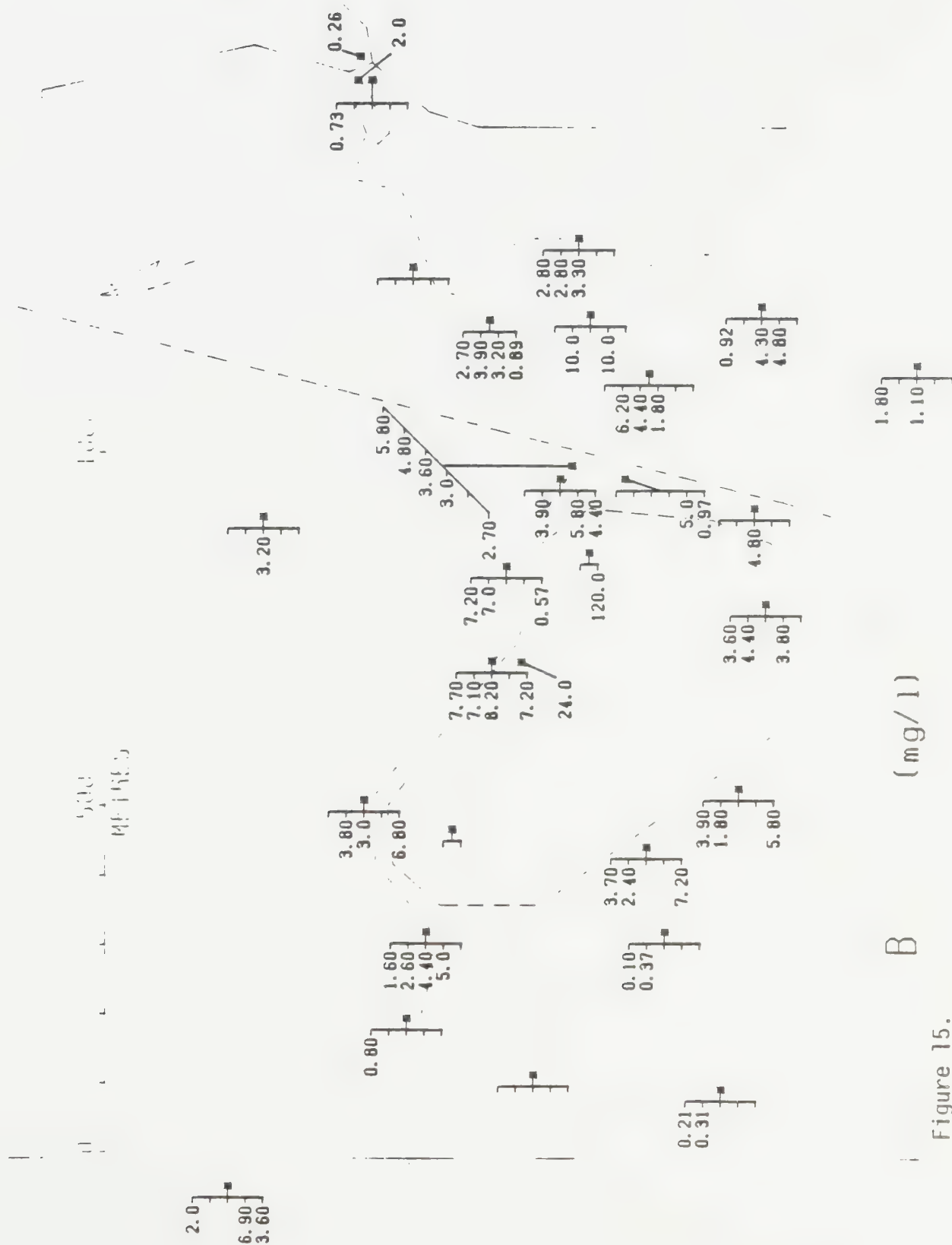
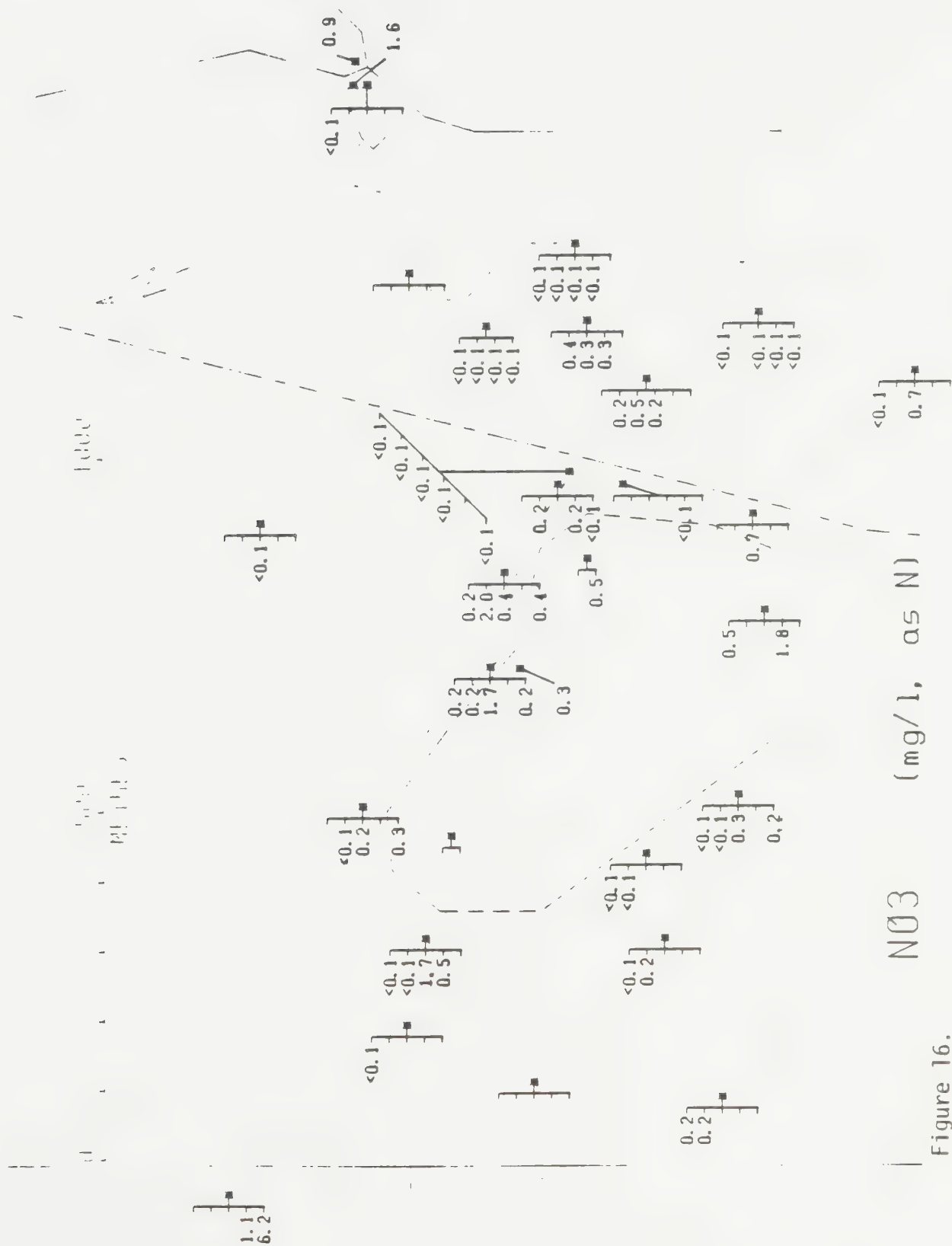
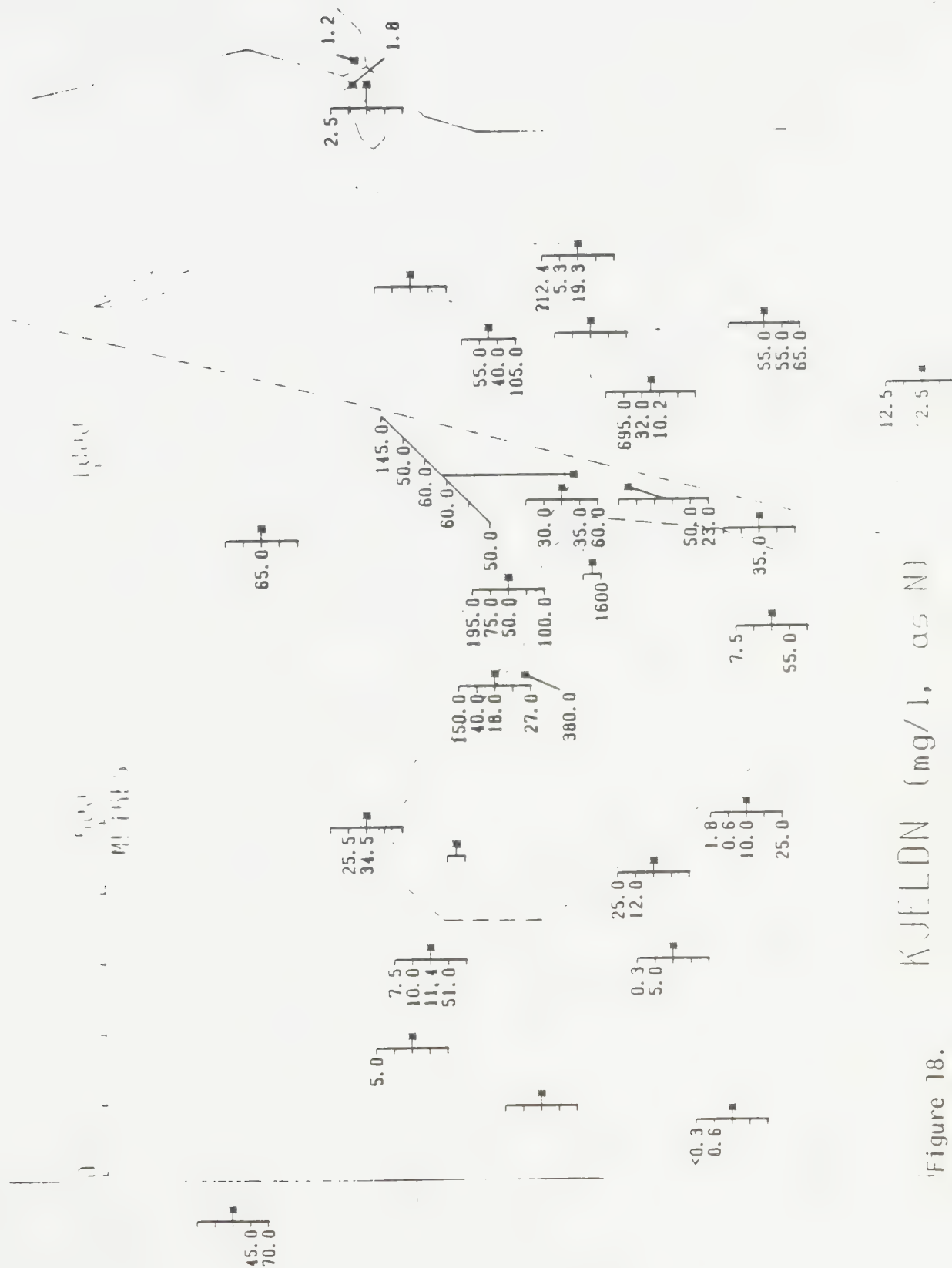


Figure 15.



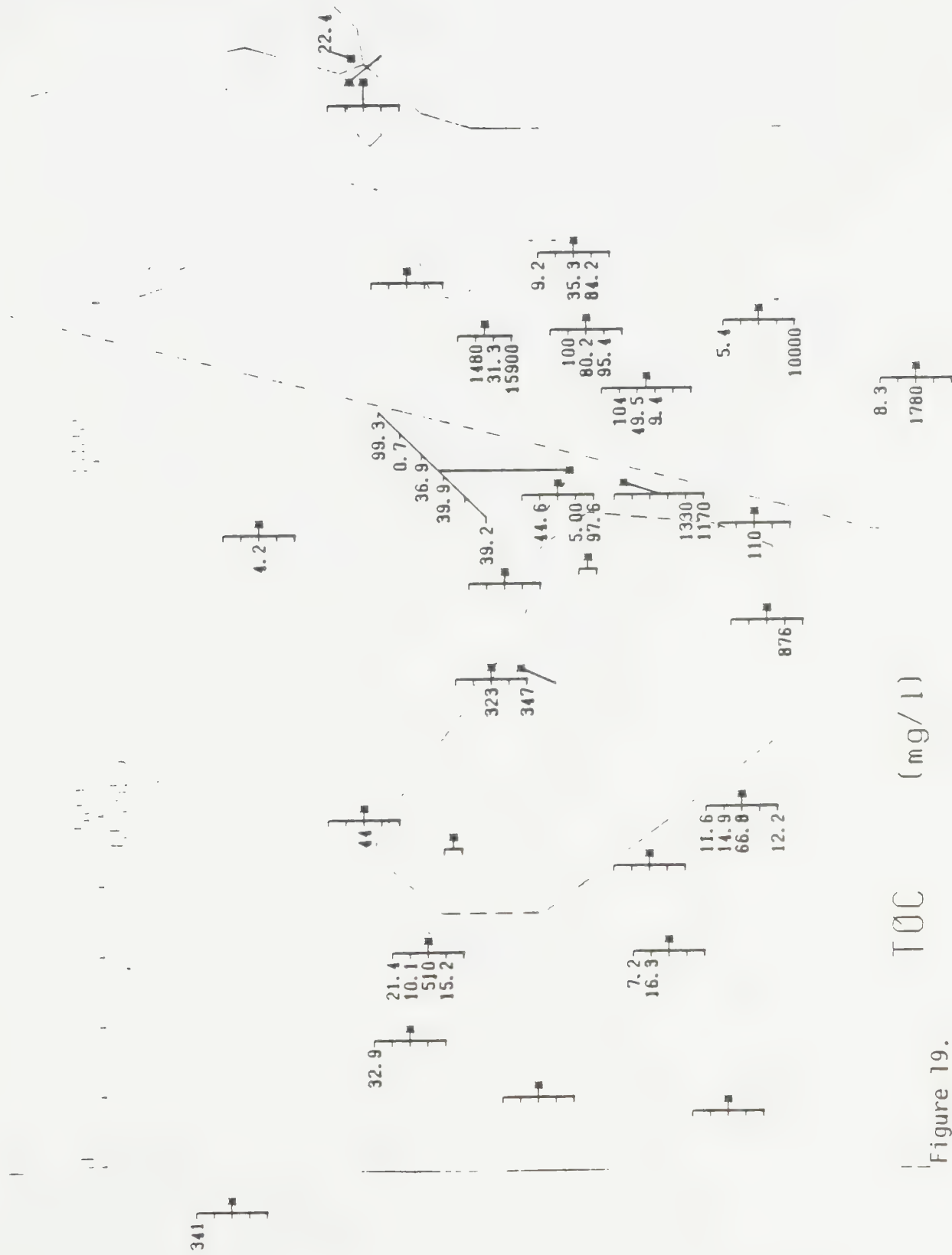
N03 (mg/l, as N)

Figure 16.



KJELDN (mg/l, as N)

Figure 18.



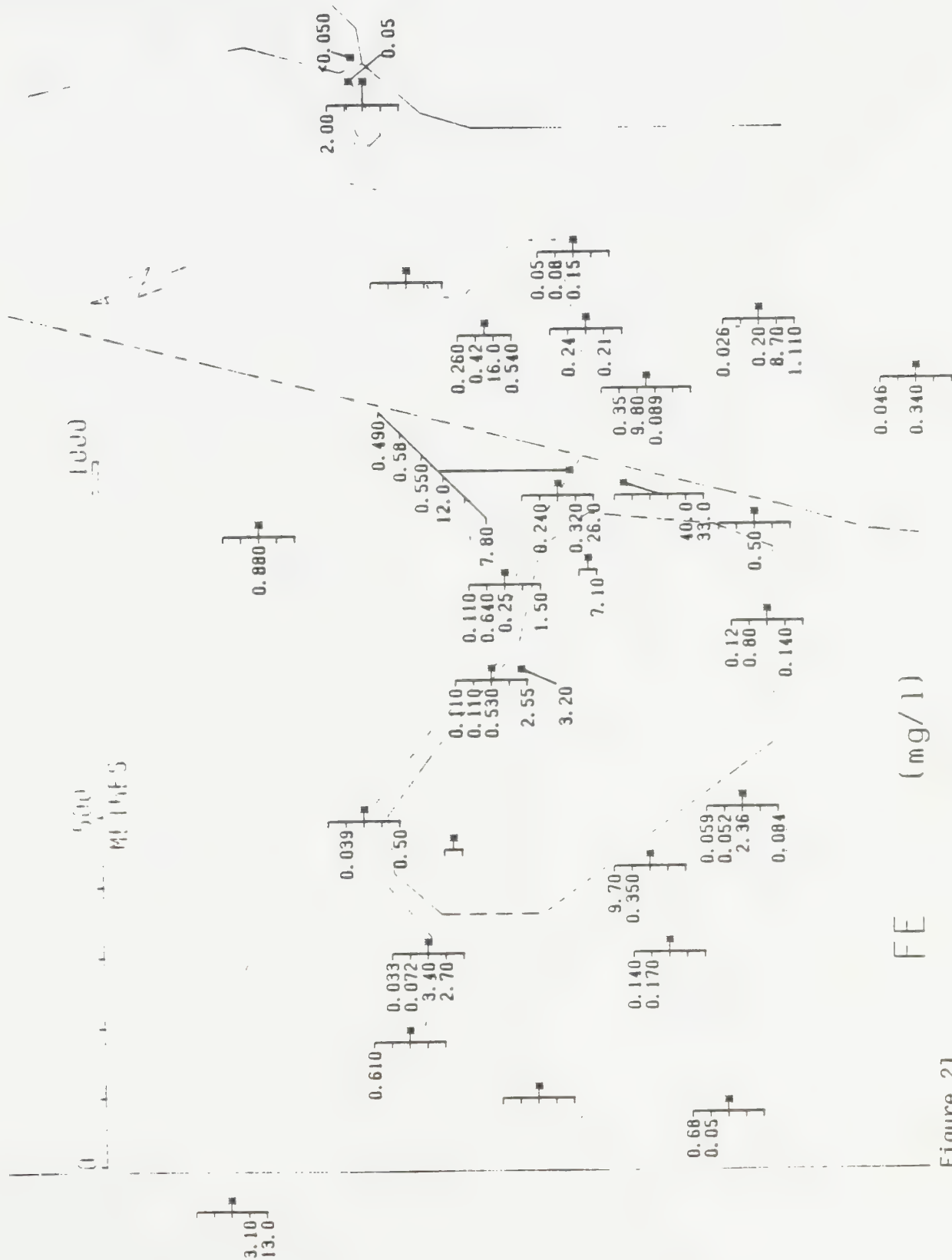


Figure 21.

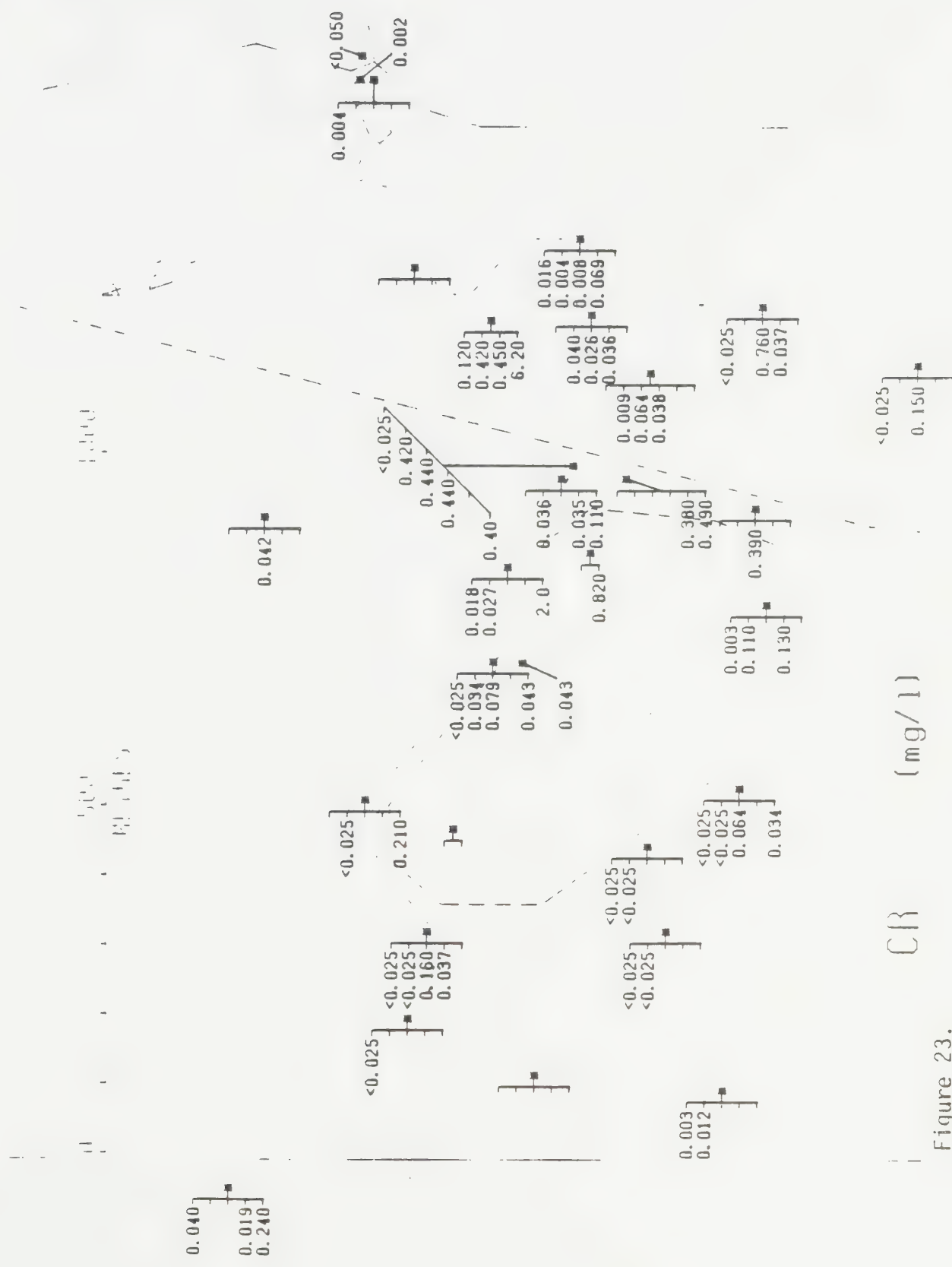
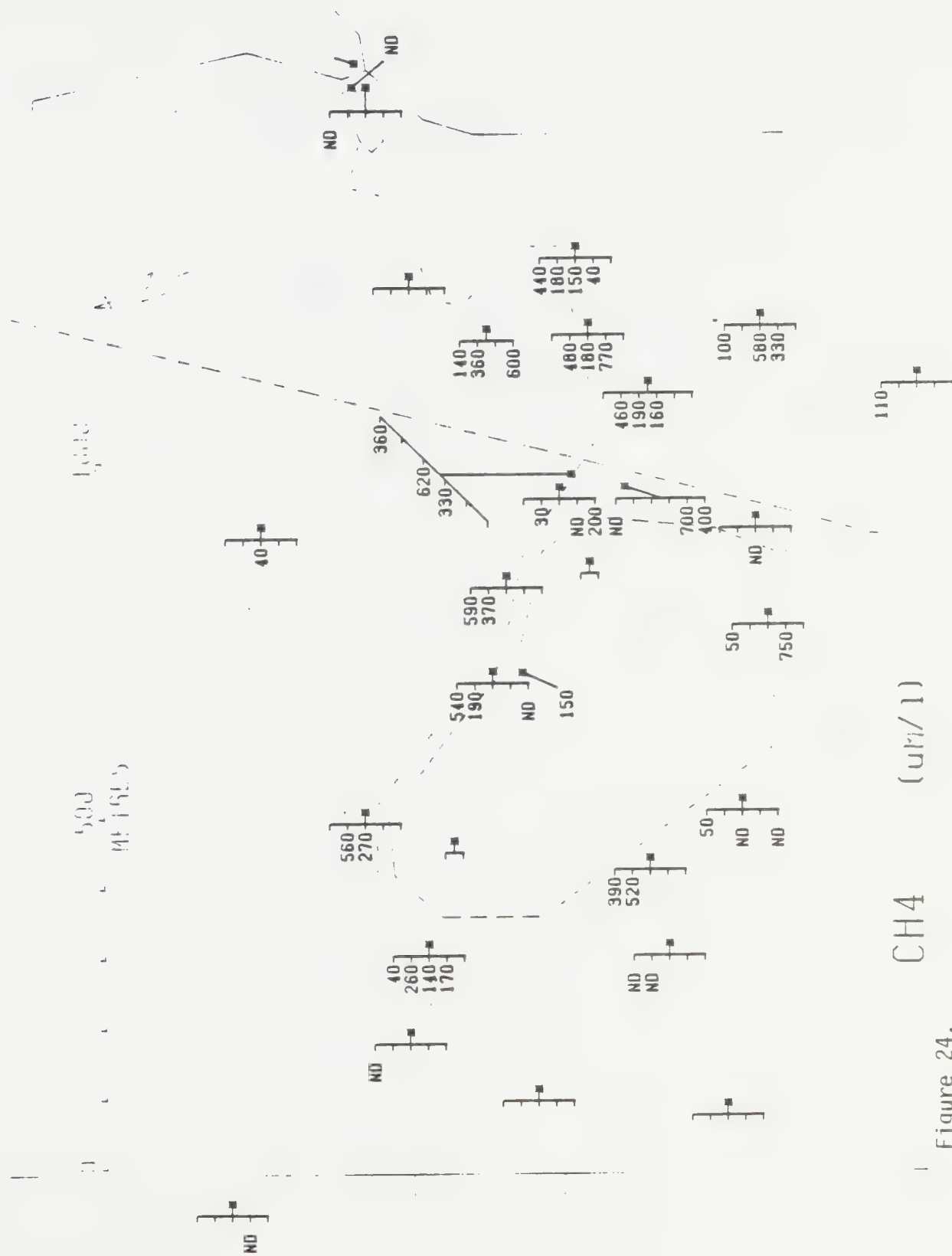


Figure 23.



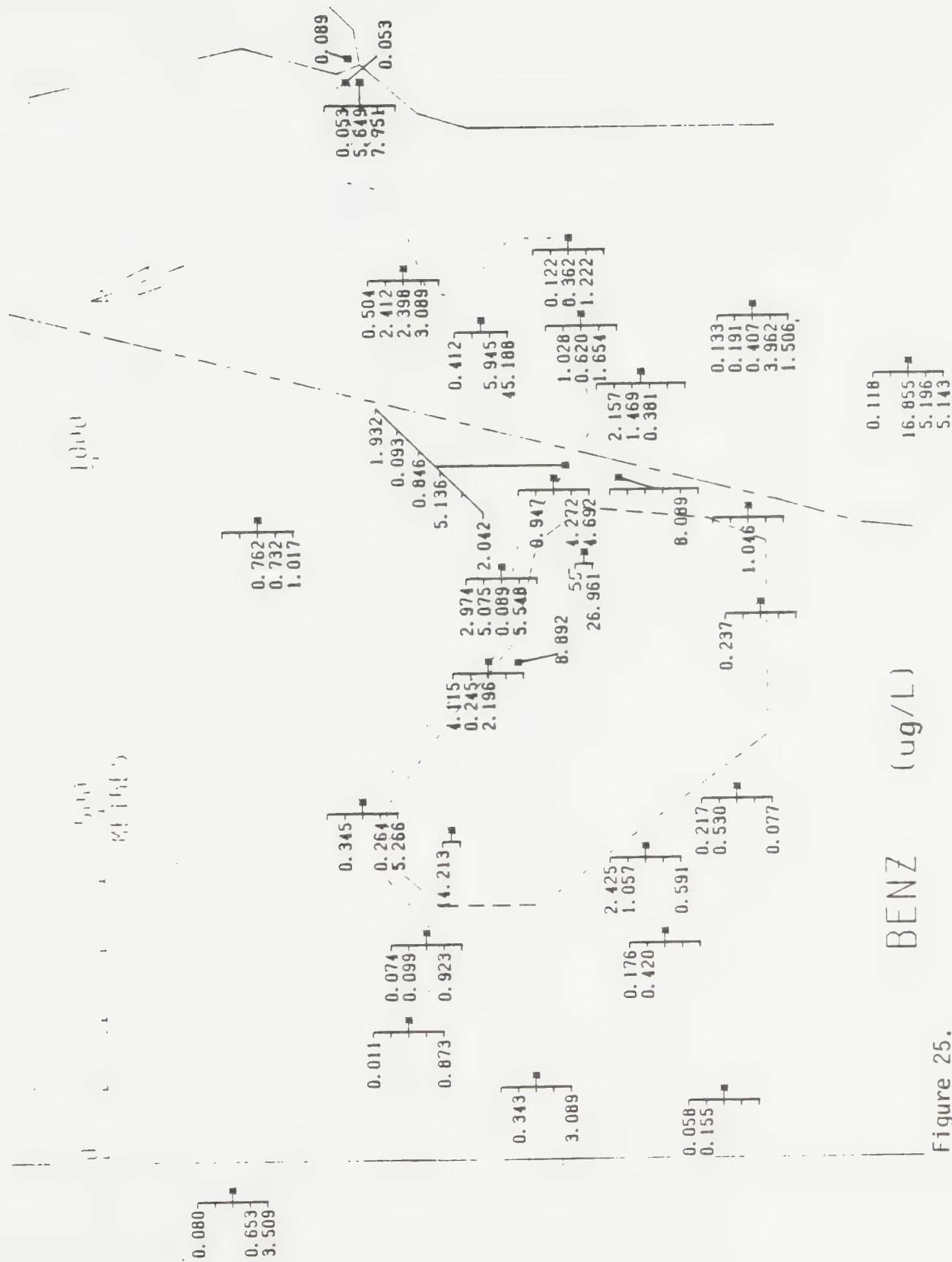
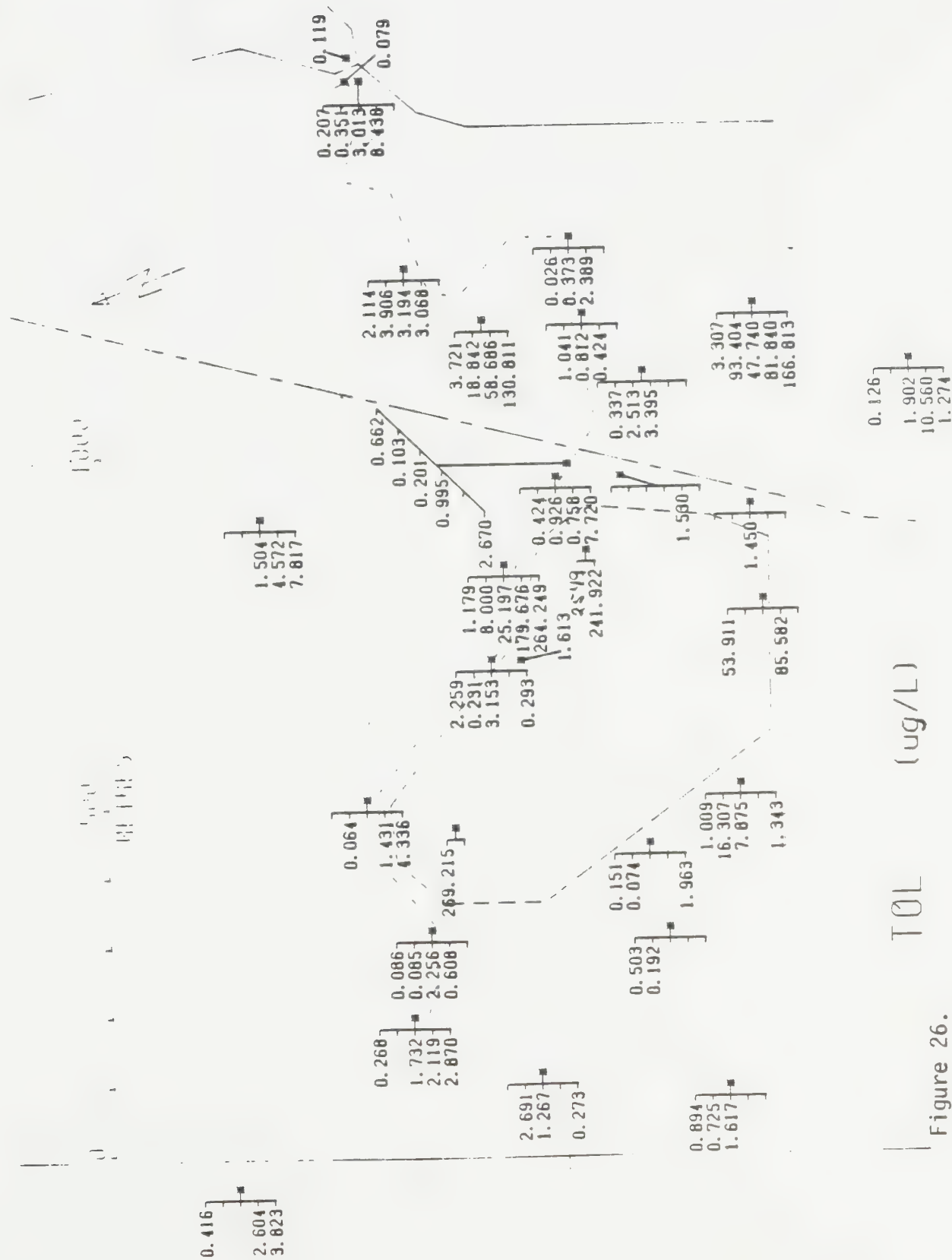
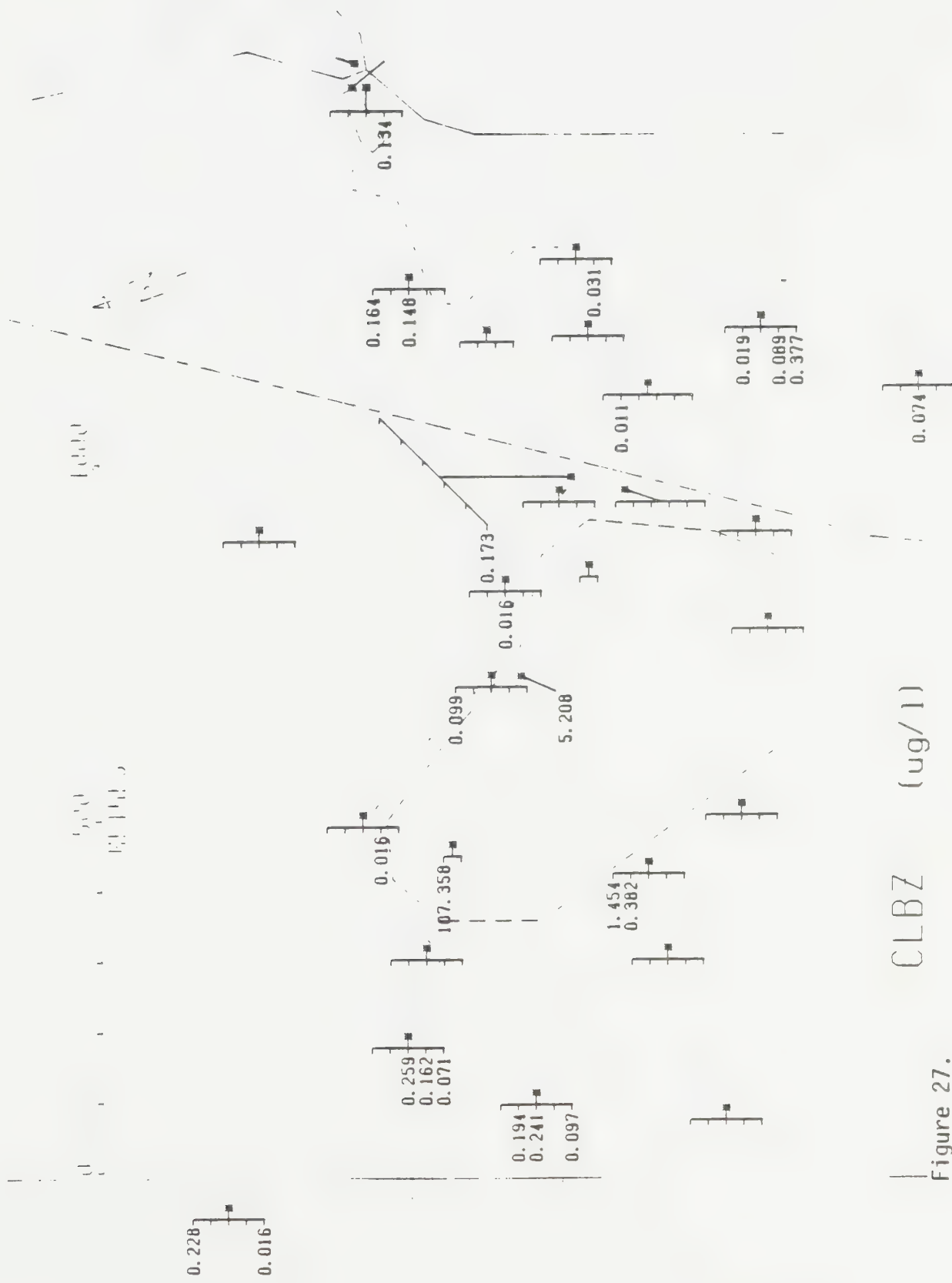


Figure 25.





CLBZ (ug/l)

Figure 27.

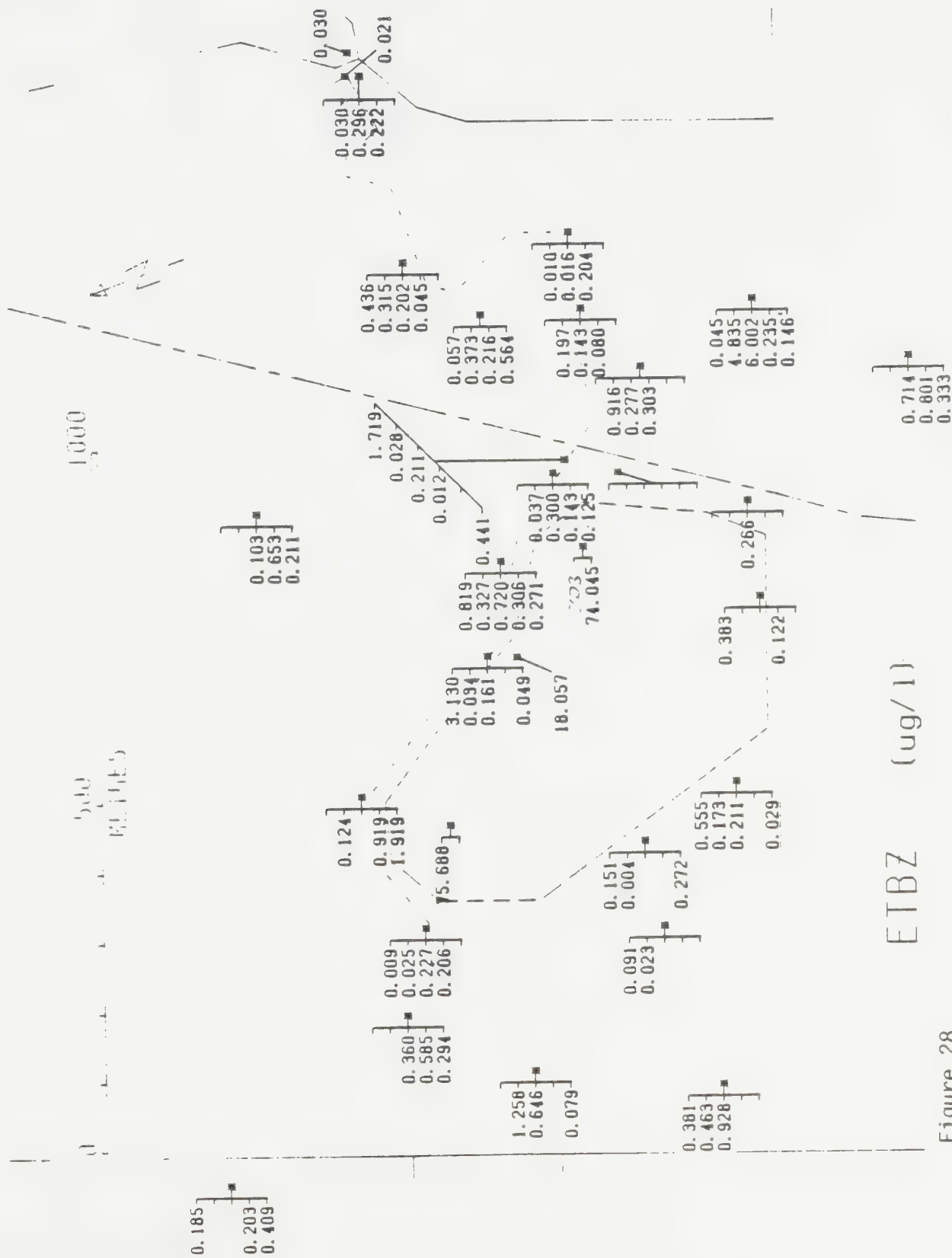


Figure 28.

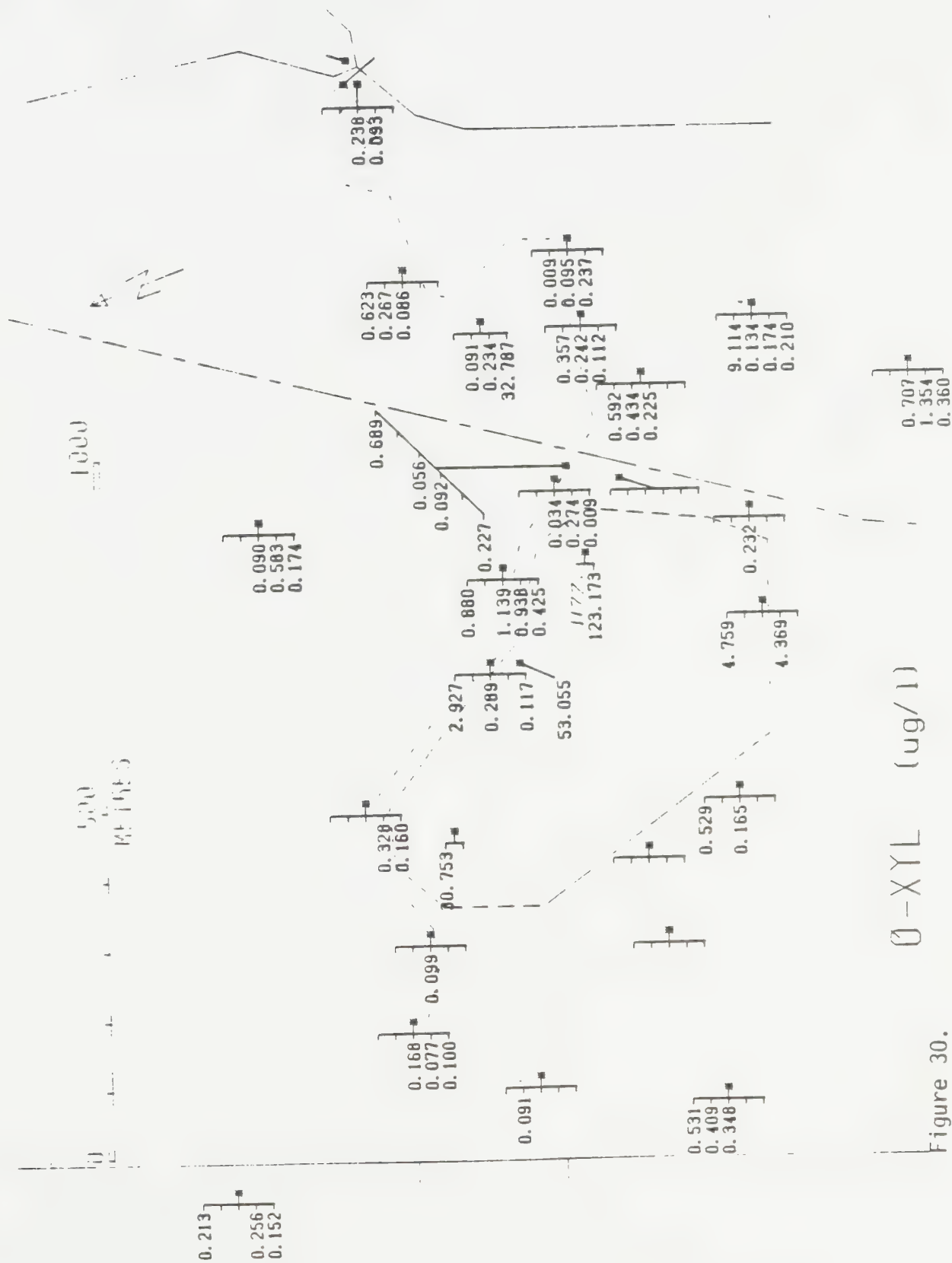


Figure 30.

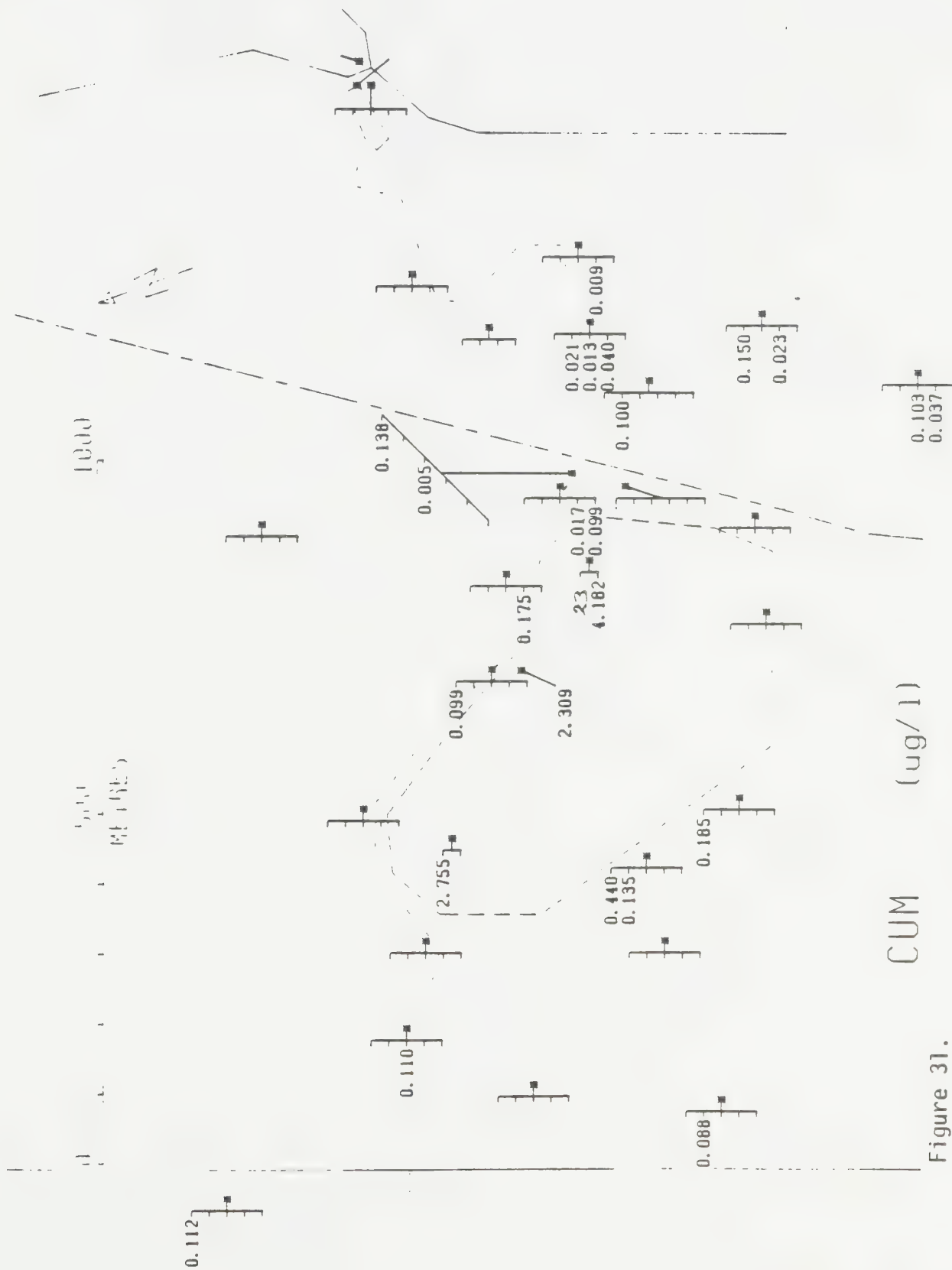


Figure 31.

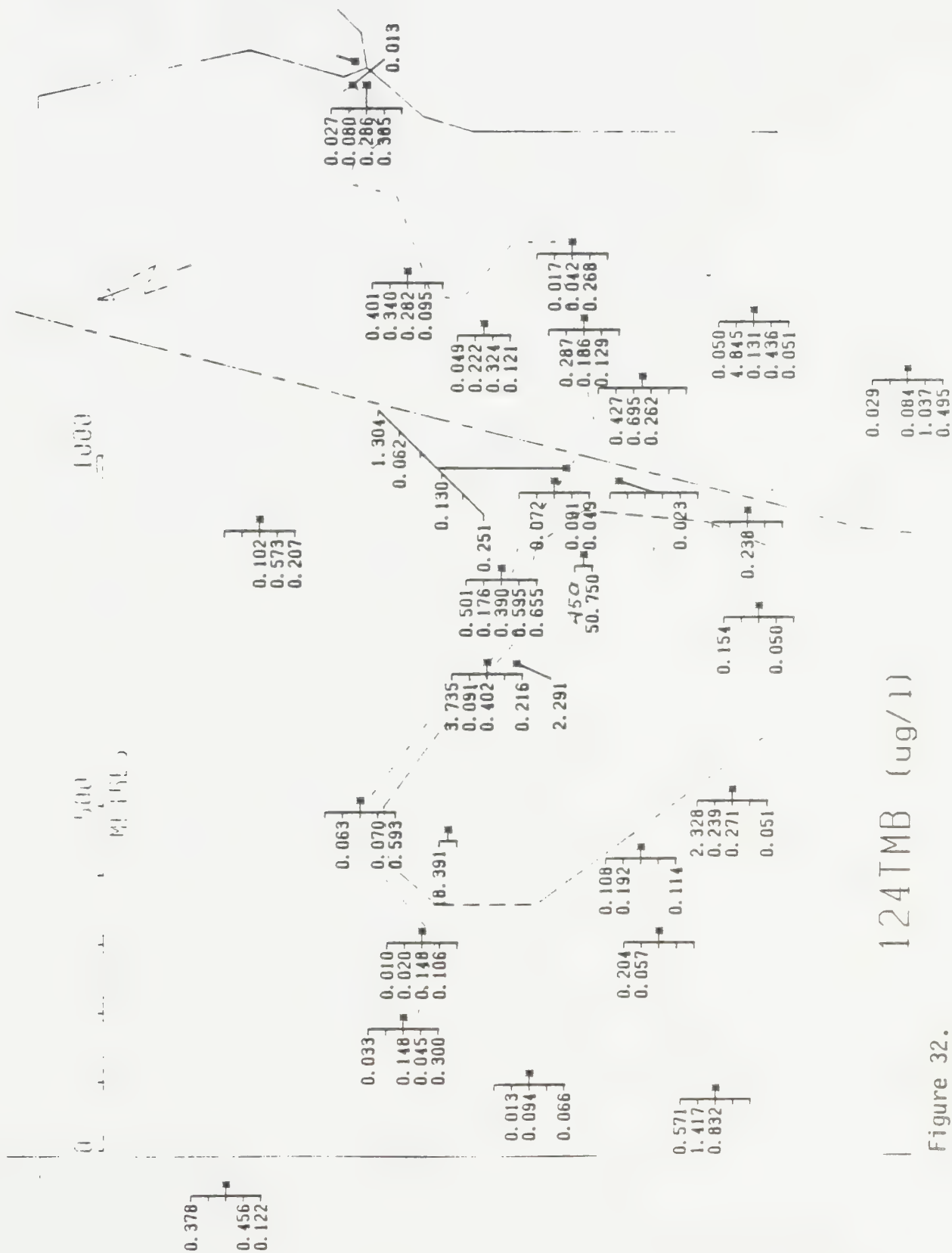


Figure 32.

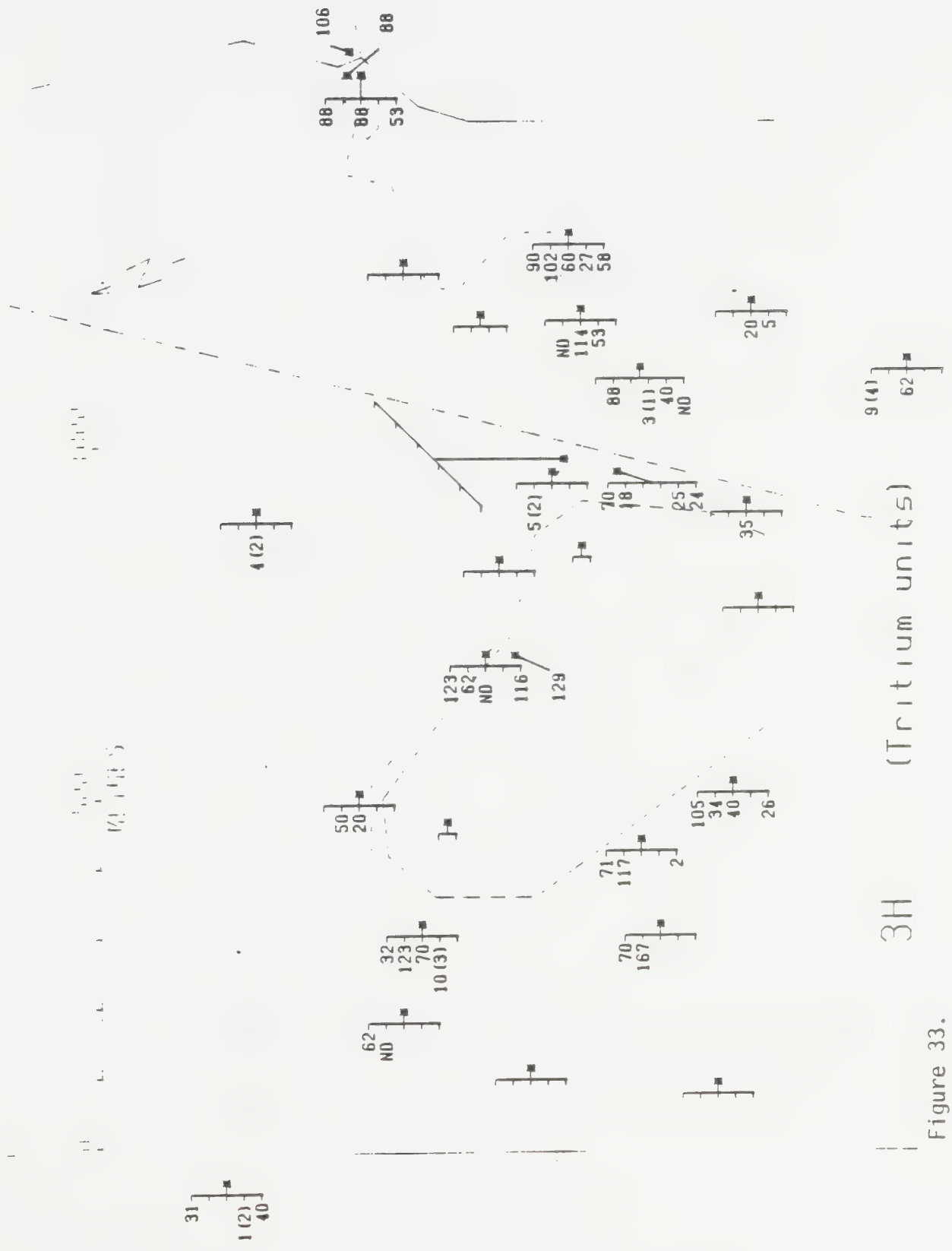


Figure 33.

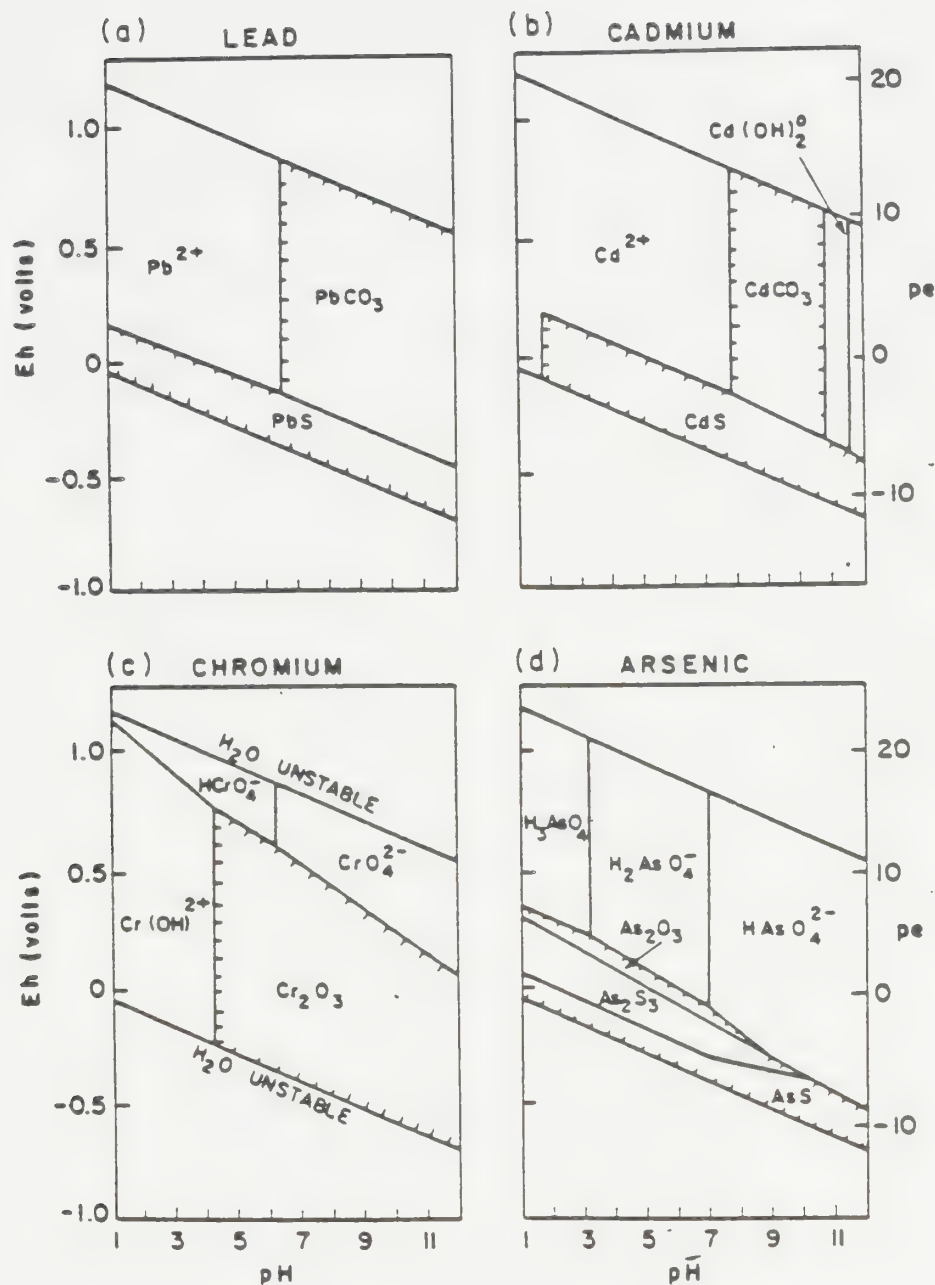


Figure 34. Eh (pe) - pH control of trace metal mobility. Conditions within hatched area favour precipitation of the phase indicated while other conditions favour the dominance of the indicated potentially mobile, aqueous species. From Cherry et al., (1984).

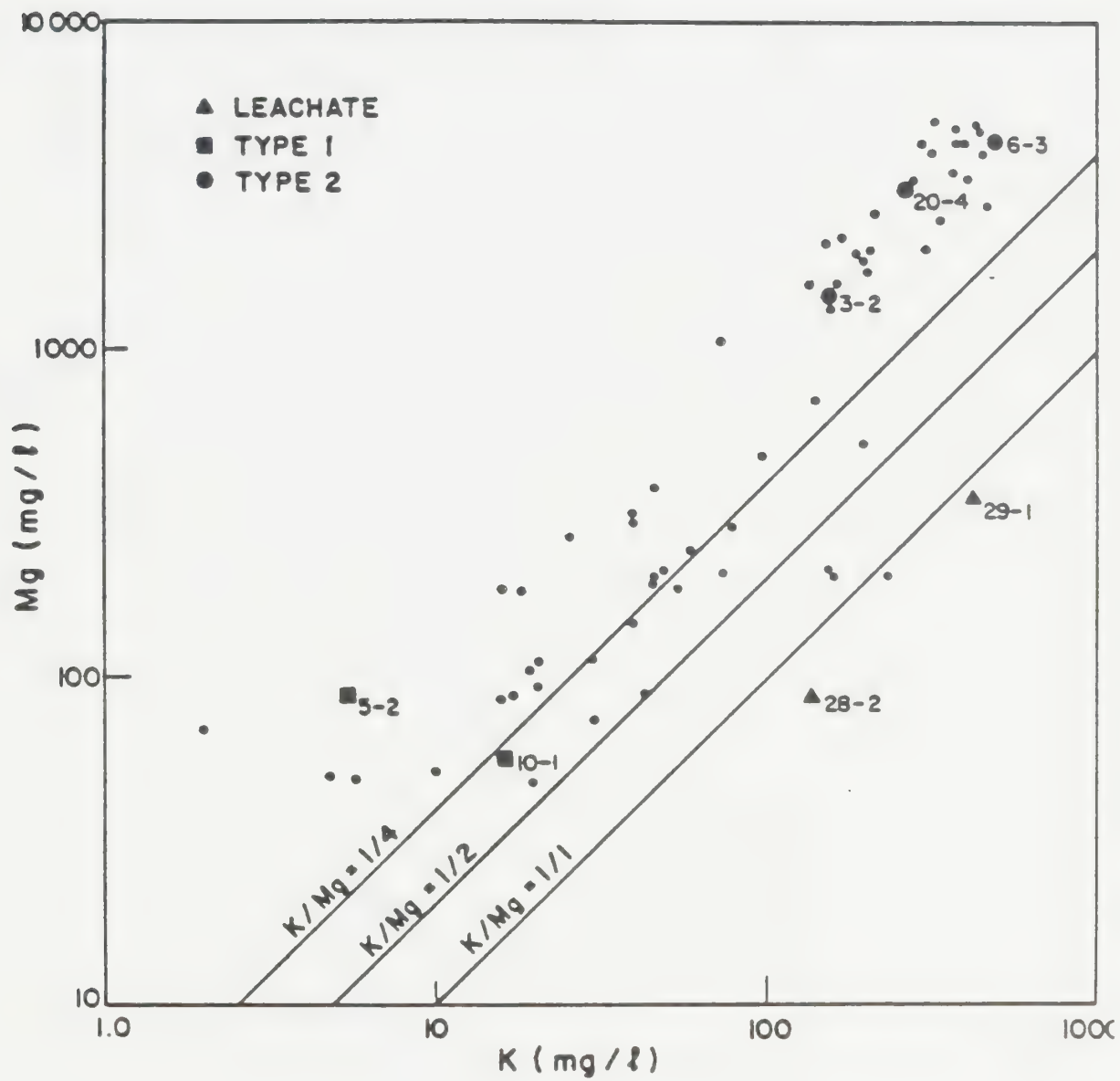


Figure 35. Relationship of potassium (K) to magnesium (Mg) in groundwater.

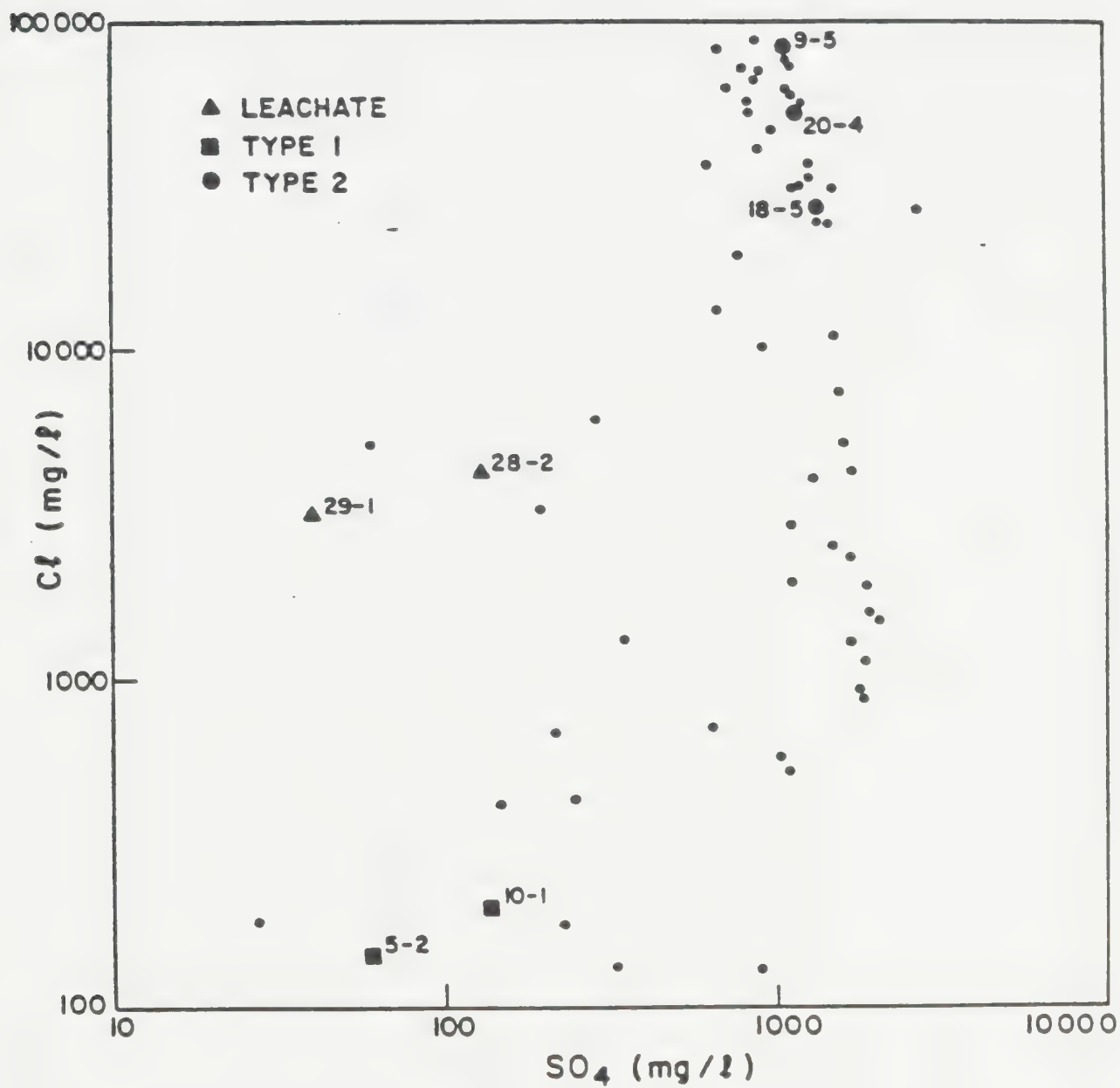


Figure 36. The relationship between sulphate (SO_4) and chloride (Cl) in groundwaters.

Table 1. Quality control procedures for chemical sampling and analyses.

PARAMETER(S)	NUMBER OF REPLICATES TAKEN ROUTINELY	WEEKLY QUALITY CONTROL MEASURES		
		BLANKS (a)	SAMPLING	REPLICATES (b)
		CHEMICAL ANALYSES		
Major Ions/Nitrogen Species/Bulk Properties	1	Glass-distilled H ₂ O, filtered in usual manner	Two, drawn consecutively from a single, acid- washed plastic reservoir	All blanks and replicates
Heavy Metals	1	Same as above	Same as above	All blanks and replicates
Purgeable Organic Compounds	2	Organic-free deionized H ₂ O	2	All blanks and at least one pair of replicates (c)
Extractable Organic Compounds	2	Organic-free deionized H ₂ O	2	All blanks and one pair of replicates
Dissolved methane and Dissolved oxygen	3 (given sufficient recharge rates)	None	3 (given sufficient recharge rates)	All replicates
Aqueous Isotopes 18O, 2H a) Conventional 3H and 3H	1	None		All replicates
b) Enriched 3H	1	None	1	All samples
Total Organic Carbon	2	None	2	All replicates; organic-free deionized water analysed daily

NOTES:

- (a) Blanks were obtained, preserved and replicated in a manner identical to that used for actual samples.
- (b) Weekly replicates were all obtained at the same sampling point whenever possible.
- (c) Inter-laboratory comparison also carried out on quadruplicate samples from eleven sampling points. See text for details.

Table 2: Inorganic species in sampling blanks (concentrations in mg/l).

LAB DATA	04/10/83 (After UW9-2)	31/10/83 (After UW22-1)
PH	6.75	6.93
COND. (US)	6.60	4.30
HARDNESS (CACO3)	<0.5	2.1
CA (mg/l)	<0.5	0.5
MG	<0.10	0.20
NA	<0.1	<0.1
K	<0.05	<0.05
CL	<0.6	<0.2
ALK., as CACO3	2.8	4.4
SO4	<0.5	<0.5
F	<0.01	/0.01
NO3, as N	<0.1	<0.1
NO2, as N	<0.01	<0.01
KJELD. N, as N	0.2	0.6
NH3, as N	<0.1	0.2
TOC	NA	NA
DOC	NA	0.3
B	0.18	0.26
FE (TOT)	0.027	0.007
MN	<0.005	<0.002
AS	<0.001	<0.001
CD	<0.005	<0.005
CR	<0.025	<0.005
CU	0.008	<0.005
NI	0.006	<0.006
PB	<0.030	<0.030
SE	<0.001	<0.001
ZN	0.033	0.011

NA - not analyzed

Table 3: Inorganic species in distilled water drawn through different filter combinations (concentrations in mg/l)

Lab Data	16/09/83 Teflon	16/09/83 Tef & Pf	16/09/83 Cell	16/09/83 Cell & Pf	16/09/83 Distill
PH	6.45	6.30	6.30	6.90	6.58
COND. (US)	2.30	4.40	4.90	4.70	2.50
HARDNESS (CAC03)	<0.5	<0.5	<0.5	<0.5	<0.5
CA (mg/l)	<0.1	<0.1	<0.1	<0.1	<0.1
MG	<0.05	<0.05	<0.05	<0.05	<0.05
NA	0.3	0.7	0.1	1.0	0.5
K	<0.05	<0.05	<0.05	<0.05	<0.05
CL	<0.2	<0.2	<0.2	<0.2	<0.2
ALK., as CAC03	3.4	5.2	2.2	4.4	2.6
S04	<1.0	<1.0	1.5	2.0	1.5
F	<0.01	<0.01	<0.01	<0.0	0.0
NO3, as N	<0.1	<0.1	<0.1	<0.1	<0.1
NO2, as N	<0.01	<0.01	<0.01	<0.01	<0.01
KJELD. N, as N	<0.2	<0.2	0.9	<0.2	<0.1
NH3, as N	<0.1	<0.1	<0.1	<0.1	<0.1
TOC	NA	NA	NA	NA	NA
DOC	1.1	1.4	1.2	0.3	0.4
B	<0.02	<0.02	<0.02	<0.02	<0.02
FE (TOT)	0.16	0.35	0.31	<0.30	0.03
MN	0.005	0.005	0.005	0.005	0.005
AS	<0.001	<0.001	<0.001	<0.001	<0.001
CD	<0.0002	<0.0002	<0.0004	<0.0002	<0.0002
CR	<0.001	<0.001	<0.001	<0.001	<0.001
CU	<0.001	<0.001	0.001	<0.001	<0.001
NI	0.002	<0.001	0.001	<0.001	<0.001
PB	<0.003	<0.003	<0.003	<0.003	<0.003
SE	<0.001	<0.001	<0.001	<0.001	<0.001
ZN	0.002	0.002	0.001	<0.002	0.001

Teflon = Teflon filter only
 Tef & Pf = Teflon & Prefilter
 Cell = Cellulose Acetate only
 Cell & Pf = Cellulose Acetate & Prefilter
 Distill = Distilled Water - unfiltered
 NA = Not analyzed

Table 6. Precision and accuracy of purgeable organic quantitations.*

Compound	Mean Concentration (n=10)		Standard Deviation (s)	Coefficient of Variation [(s/ \bar{x})x100%]	Calculated Concentration (μ)	t(= $\frac{\sqrt{10}(\bar{x}-\mu)}{s}$)	P(t=0)
	(\bar{x})	(μ)					
benzene	4.123 μ g/l	4.123	0.384	9.3%	4.040	0.684	P>0.5
toluene	3.947	3.947	0.351	8.9%	3.960	0.117	P>0.5
chlorobenzene	5.786	5.786	0.487	8.4%	5.708	0.507	P>0.5
ethylbenzene	3.868	3.868	0.432	11%	3.900	0.234	P>0.5
(p+m)-xylene	3.825	3.825	0.367	9.6%	3.884	0.508	P>0.5
o-xylene	4.077	4.077	0.374	9.2%	4.072	0.0423	P>0.5
isopropyl benzene	3.885	3.885	0.419	11%	3.972	0.657	P>0.5
1,2,4-trimethylbenzene	4.037	4.037	0.396	9.8%	4.120	0.663	P>0.5
naphthalene	3.663	3.663	0.529	14%	5.248	9.47	P<0.001

*Based on ten replicate analyses of a single standard mixture.

Table 7. GC/MS confirmation of peaks identified by GC/FID-purgeable organics. Concentrations are in $\mu\text{g/l}$ (a)

Sample Number (Replicate)	Analytical Laboratory (b)	Column Used (c)	benzene	toluene	chloro-benzene	ethyl-benzene	p- + m-xylene	o-xylene
UW 7-1 (B) (D)	UW (=RD) MTL (=CC)	DB-5 Superox	4.115 3.532	2.259 1.30	0.099 - (d)	3.130 3.089	6.668 p:3.39 m:3.36 } 6.75	2.927 2.77
UW 12-3 (D) (A)	UW MTL	DB-5 Superox	1.046 1.29	1.450 2.23	- -	0.266 0.81	0.745 p:0.96 m:2.02 } 2.98	0.232 1.14
UW 27-6 (A) (B) (C)	UW UW MTL	DB-5 DB-5 Superox	0.727 1.000 0.75	18.665 9.398 8.21	- 0.031	0.189 0.143 0.35	0.640 0.524 p:0.47 m:1.05 } 1.52	0.366 0.175 0.51
UW 22-1 (B) (D)	UW MTL	DB-5 Superox	2.974 2.270	1.179 0.8242	- 0.0430 (e)	0.819 0.656	1.830 p:0.600 m:0.769 } 1.37	0.880 0.9190
UW 22-2 (B) (D)	UW MTL	DB-5 Superox	5.075 4.941	8.000 9.021	- -	0.327 0.4229	0.359 m: 0.314	- -
UW 28-2 (A) (B)	UW MTL	DB-5 Superox	26.96 7.158	241.9 99.59	- 0.288 (e)	74.05 74.12	191.1 p: 72.5 m:>125.9 } >198 (f)	123.2 >126.5 (f)
UW 29 (D) (Seep near UW 7) (A)	UW MTL	DB-5 Superox	8.892 0.5800	1.613 -	5.208 0.228	18.06 -	12.52 p:1.868	53.06 9.798
UW 8-5 (A) (B)	UW MTL	DB-5 DB-5	5.266 (0.94) (g)	4.336 3.814	- -	1.919 2.09	1.301 p:2.29 m:2.14 } 4.43	0.160 0.268
UW 16-4 (A) (B)	UW MTL	DB-5 DB-5	0.732 -	4.572 5.22	- -	0.653 0.587	1.398 1.71	0.583 0.634
UW 17-1 (A) (B)	UW MTL	DB-5 DB-5	2.425 2.555	0.151 0.205	1.454 1.599	0.151 0.282	0.875 1.097	- -
UW 19-4 (A) (B)	UW MTL	DB-5 DB-5	7.751 -	8.438 4.82	0.134 -	0.222 0.182	0.821 0.4962	0.093 0.216

Table 8. Purgeable organics identified by GC/MS. Concentrations are in $\mu\text{g/l}$. Otherwise, relative abundance is indicated, where \checkmark is $<1\%$, + is 1 to 10% and M is $>10\%$ of detector response.

COMPOUND	3-1	6-1	7-1	8-5	10-1	12-3	14-4	14-5	16-4	19-1	19-4	22-1	22-2	24-4	27-6	28-2	29-1
	Sta	Sta	Sta		Sta		Sta			MOE			MOE	MOE	MOE	MOE	Sta
<u>Halogenated Hydrocarbons</u>																	
dichloromethane (methylene chloride)(b)			6.1	3.3		5.6	\checkmark	\checkmark				1.6	3.3	40			4.8
chloroform (a)	\checkmark	\checkmark	2.8		\checkmark		0.4	\checkmark		0.6			1.2	1.6	0.5	1.0	1.5
bromoform	\checkmark				\checkmark												0.3
1,1-dichloroethane			9.6									3.3		5.8			
1,2-dichloroethane													3.5				
1,1,1-trichloroethane (a)	\checkmark	\checkmark	0.1		\checkmark		0.2						3.6	19	9.2		0.1
chloroethylene (vinyl chloride)																\checkmark	
1,2-dichloroethylene												2.4				10	
tetrachloroethylene	\checkmark	\checkmark	\checkmark		\checkmark												
1-chlorobutane								\checkmark									
chlorobenzene	\checkmark	\checkmark			\checkmark		0.3									-	4.0
1,3-dichlorobenzene		\checkmark					0.1										
1,4-dichlorobenzene	\checkmark	0.08			\checkmark		0.4										3.3
<u>O,N,S Containing</u>																	
1,1'-oxybis-ethane				+													\checkmark
2,2'-oxybis propane																	\checkmark
acetone (b)				+				\checkmark		2.0		1.2	1.7	5.6	1.4		\checkmark
1-phenylethanone																	\checkmark
1-methylphenylethanone																	\checkmark
2-butanone (MEK) (b)						+	+			1.1		\checkmark	120	+	2.6		\checkmark
3-methyl-2-butanone																	\checkmark
3-pentanone (diethyl ketone)																	\checkmark
2,4-dimethyl-3-pentanone																	\checkmark
4-methyl-2-pentanone (MIBK)																	\checkmark
2-hexanone																10	
3-hexanone									\checkmark								
5-methyl-2-hexanone									\checkmark								\checkmark
3,3,5-trimethylcyclohexanone													\checkmark				
3-heptanone									\checkmark				\checkmark			88	
fenchone																44	\checkmark
camphor													\checkmark				\checkmark
3-methyl-4-heptanone													\checkmark				\checkmark
ethanol								\checkmark									
(R)-2-butanol						+											
1,3-butanediol																	
2-methyl-2-propanol (t-butanol)																	
4-penten-2-ol												\checkmark				29	
4-ethyl-1-octyn-3-ol																	
4-methyl-3-pentenoic acid																	
tetrahydrofuran (THF) (b)								\checkmark		6.0			170	>270	+	24	\checkmark
1,3-dioxane																1.0	
1,4-dioxane														0.8		27	\checkmark
2 or 4-methyl-1, 3-dioxolane																	
1,3-epoxy-4-methylpentane																	
2,3-dimethyloxirane																	
1,3-oxathiolane																	
2-methylthiophene						\checkmark										14	
sulphur dioxide						\checkmark		\checkmark									
carbon disulphide			+														
dimethyl disulphide								\checkmark					0.7	2.2			
diethyl disulphide				\checkmark									3.9	0.6			
ethyl cyanide																1.1	
<u>Hydrocarbons</u>																	
propane						\checkmark					\checkmark						
1-butene																	
pentene															8.1	\checkmark	
3-methylpentane										0.8							
2,2,3,4-tetramethylpentane																	
1-methyl-4-isopropenylcyclohexene																	\checkmark
C ₇ H ₁₆										1.6			2	1.4		37	
C ₈ H ₁₈										1.6					1.4	6	
C ₁₀ H ₂₂																	
benzene	0.4	0.9	3.5	\checkmark	0.3	1.3	0.6	\checkmark				2.2	25	18			
toluene	0.8	0.9	1.3	3.8	\checkmark	2.2	0.7		5.2	1.8	4.2	0.8	4.9		0.8	7.2	0.6
ethylbenzene	\checkmark	0.08	3.1	2.1	\checkmark	0.8	0.2						9.0	>85	8.2	100	1.3
xylene	0.3	0.3	9	5	0.1	4	1.2	\checkmark	0.6	0.2	0.7	0.4	0.4	0.6	0.4	74	24
propylbenzene	\checkmark	\checkmark	\checkmark		\checkmark				2	1.4	0.7	2.3	0.3	3	2	>320	160
cumene																	21
substituted benzenes																	
C ₉ H ₁₂	0.1	0.4	4	\checkmark	\checkmark		1.1									23	1400
C ₁₀ H ₁₆																	
C ₁₁ H ₁₈																	

- (a) Detected in leach test of piezometer material (Goodman, 1983, Appendix E)
 (b) Detected in leach test of glued PVC (Boettner et al., 1982, or PVC cement (Lockwood and Priddle, 1983).

Table 9. Extractable organics in base/neutral fraction. Relative abundance is indicated where ✓ is >10%, † is 1 to 10% and M is >10% of total detector response. Analyses by Stanford University (Sta) and Ontario Ministry of the Environment (MOE) are indicated

COMPOUND	1-2	2-4	3-4	6-1	7-1	7-3	10-1	12-3	14-1	17-2	17-5	18-3	19-1	19-5	20-4	20-5	21-3	21-4	22-2	23-1	23-4	24-4	27-1	27-4	27-6	28-2	29-1	31-1	
	Sta	Sta	Sta	Sta	Sta	Sta	Sta	Sta	Sta	Sta	MOE	MOE	MOE	MOE	MOE	MOE	MOE	MOE	MOE	MOE	MOE	MOE	MOE	MOE	MOE	MOE	Sta	MOE	
<u>HYDROCARBONS</u>																													
<u>General</u>																													
<u>C₉</u>																													
<u>Polycyclic Aromatics HC</u>																													
Naphthalene*	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	
Phenanthrene*																													
Acenaphthene*																													
Fluoranthene*																													
Biphenyls																													
<u>HETEROCYCLIC COMPOUNDS</u>																													
Octamethyl-cyclotetrasiloxane																													
1,3-dioxane, derivatives																													
Oxadiazolin, substituted	✓																												
Pyrrolidinone, substituted																													
2,4 dimethyl oxazole																													
Benothiazole and subst. (b)																													
1,3,5-trithiane																													
1,2,4-trithiolane and subst. (a)																													
Hexathiepane																													
<u>Anides</u>																													
Nonanamide																													
Aniline																													
-- , N-methyl																													
-- , N,N-dimethyl																													
<u>Ethers</u>																													
2-butoxy-ethanol																													
2-(2-ethoxyethoxy)-ethanol																													
2,2-oxybis-ethanol (b)																													
(diethylene glycol)																													
<u>ESTERS</u>																													
Butanoic acid,																													
1-methyl propyl ester																													
Propanoic acid, propyl ester																													
-- , 2-methyl-																													
propyl ester																													
Hexanedioic acid, di-octyl ester																													
Phosphoric acid, triethyl ester																													
Phosphoric acid, tributyl ester (a)																													
Diethyl-phthalate* (a,c)																													
Di-n-butyl phthalate* (c)																													
Di-(cyclohexyl)-phthalate																													
Di-isooctyl-phthalate																													
Bis-(2-ethylhexyl phthalate* (a,c)																													
<u>Alcohols</u>																													
2-ethyl-4-methyl-1-pentanol																													
<u>PCBs</u>																													
<u>Others</u>																													
Molecular sulphur (S ₈)																													

* Included in U.S. EPA list of priority pollutants (USEPA, 1977). (a) Detected in piezometer material leach test (Goodman, 1983, Appendix E), (b) Major component in leach test a) above (c) Detected in PVC leach test by Junk et al., 1974.

Extractable organics in acid fraction. Relative abundance indicated where \checkmark is $<1\%$, $+$ is 1 to 10% and M is $>10\%$ of total detector response. Analyses by Stanford University (Sta) and Ontario Ministry of the Environment (MOE) are indicated.

COMPOUND	1-2	2-4	3-4	6-1	7-1	7-2	7-3	10-1	12-3	13-5	14-1	17-5	18-1	18-3	19-1	19-5	20-5	21-3	22-2	23-2	23-4	24-4	27-1	27-4	27-6	28-2	29-1	31-1	
	Sta	Sta	Sta	Sta	Sta	Sta	Sta	Sta	Sta	Sta	Sta	Sta	Sta	Sta	Sta	Sta	Sta	Sta	Sta	Sta	Sta	Sta	Sta	Sta	Sta	Sta	Sta	Sta	
Carboxylic Acids																													
C ₄ butanoic, 3,3-dimethyl																													
C ₅ pentanoic (valeric)																													
C ₆ hexanoic																													
C ₈ octanoic (b)																													
C ₉ nonanoic																													
C ₁₀ decanoic (capric) (b)																													
C ₁₂ dodecanoic (lauric) (c)																													
C ₁₆ hexadecanoic (palmitic)																													
C ₁₈ octadecanoic (stearic) (c)																													
Aromatic Acids																													
benzoic (b)																													
2-hydroxybenzoic (salicylic) (a)																													
4-(1,1-dimethylethyl)-benzoic																													
benzeneacetic																													
Ethers																													
2-butoxy ethanol																													
2-(2-ethoxyethoxy) ethanol																													
2-ethoxy-2-methyl propane																													
2-2'-oxybis-ethanol (b)																													
Esters																													
butyl methyl phthalate																													
butyl-(2-methylpropyl) phthalate																													
di-butyl phthalate * (a,c)																													
di-isooctyl phthalate																													
bis-(2-ethyl hexyl)																													
phthalate (a,c)																													
Heterocyclics																													
benzothiazole and subst. (b)																													
1,2,4-trithiolane (a) and subst.																													
hexathiepane																													
Others																													
dilodmethane																													
molecular sulphur																													
phenols (a)																													
dimethyl phenol																													
cresols (b)																													

* Included in the US EPA list of Priority Pollutants (1977)
 (a) detected and (b) a major component in piezometer material leach test (Goodman, 1983, Appendix C)
 (c) detected in PVC leach test, Junk et al (1974)

Table 12. Possible type 2 background groundwaters and a landfill leachate (29-1) for comparison.

Sample	3-2	16-3	20-4	29-1 (seep)
Date	12/09/83	06/10/83	16/05/83	27/10/83
Tritium (T.U.)	n.d.	2	2	129
pH	6.70	6.06	6.70	7.67
Cond. (uS)	64000	150000	112000	13100 -
Ca (mg/l)	5500	16400	11500	87.5
Mg	1460	4300	3150	355
Na	8700	29400	18400	2120
K	156	490	300	408
Cl	27300	84600	61400	3190
Alk (as CaCO ₃)	298	4510	----	3540
SO ₄	1330	1080	1200	40.0
F	----	----	----	1.3
NO ₃ as N	0.4	<0.1	<0.1	0.3
Kjeld. N	----	65.0	>39.3	380
DOC/TOC	100	4.2	36.6	294
Fe	0.24	0.88	1.2	3.2
Mn	0.35	1.10	1.6	0.20
As	0.001	<0.001	<0.001	0.003
Cd	0.005	<0.005	<0.010	<0.005
Pb	<0.030	0.086	<0.060	0.032
Se	<0.001	<0.001	0.001	0.001
Zn	0.060	0.036	0.079	0.028 -
Date (organics)	13/09/83	06/10/83	24/10/83	27/10/83
Benzene (ug/l)	1.03	0.76	0.65	8.9
Toluene	1.04	1.50	2.50	1.6
Cl-benzene	n.d.	n.d.	n.d.	5.2
Ethylbenzene	0.20	0.10	0.20	18.1
p-Xylene	0.55	0.14	0.67	12.5
o-Xylene	0.36	0.09	0.26	53.1
Napthalene	1.95	0.10	0.27	10.7
n.d. - not detected				

Figure 13. Landfill leachate chemistry and MOE water quality criteria for public water supplies.

Sample Date	29-1 (seep) 27/10/83	28-2 01/11/83	26-1	BH 2-2 ^a 04/05/80	MOE water quality criteria
pH	7.67	7.86		7.3	
Cond. (uS)	13100	25900		17500	
Ca (mg/l)	87.5	19.0			
Mg	355	86.0			
Na	2120	4700			
K	408	135			
Cl	3190	4360		2750	250
Alk (as CaCO ₃)	3540	9130			
SO ₄	40.0	130			250
F	1.3	6.6			
NO ₃ -N	0.3	0.5			10
Kjeld. N	380	1600		1760	
DOC/TOC	294	4600			
Fe	3.2	7.1		21.0	0.3
Mn	0.2	0.16			
As	0.003	0.029			
Cd	<0.005	<0.01		<0.02	0.01
Pb	0.032	0.097		0.5	0.05
Se	0.001	0.004			
Zn	0.028	0.47		2.5	5
<hr/>					
Date (organics)		11/11/83	14/12/83		
Benzene (ug/l)	8.9	27.0	14.2		
Toluene	1.6	242	269		
Cl-benzene	5.2	n.d.	107		
Ethylbenzene	18.1	74.0	75.7		
p-Xylene	12.5	191	76.3		
o-Xylene	53.1	123	30.7		
Napthalene	10.7	60.3	131		

n.d. - not detected

a - from a report by Gartner-Lee Assoc. Ltd., project 79-78, Sept. 1980

Table 14. Landfill leachate impact "scores" and the number of geochemical tests applied.

Piezometer	No. of Tests	Score (%)	Piezometer	No. of Tests	Score (%)	Piezometer	No. of Tests	Score (%)
1-2	12	17	10-1	11	14	21-1	5	0
-3	5	0	-3	2	25	-2	2	50
-4	4	0	-4	2	25	-3	12	25
			-5	2	0	-4	10	10
2-1	3	0				-5	5	60
-2	5	0	11-2	2	25			
-3	5	10	-3	2	25	22-1	7	50
-4	11	9	-5	2	25	-2	12	25
						-3	5	40
3-1	2	25	12-3	12	21	-4	2	50
-2	5	0				-5	5	40
-3	5	0	13-5	10	30			
-4	9	11	-6	3	0	23-1	9	6
						-2	10	10
4-1	2	25	14-1	9	11	-3	5	10
-2	2	0	-3	5	30	-4	11	36
-3	2	25	-4	4	38			
-4	2	0	-5	4	13	24-1	5	10
						-4	14	43
5-1	5	0	15-1	5	0			
-2	5	0	-2	5	10	26-1	2	100
-3	2	0						
			16-3	5	0	27-1	12	25
6-1	5	40	-4	4	0	-2	5	0
-2	5	0	-5	2	0	-3	5	0
-3	3	0				-4	12	21
-4	5	0	17-1	6	67	-6	12	46
-5	7	14	-2	8	19			
			-5	7	36	28-1	2	0
7-1	13	42				-2	13	54
-2	9	6	18-1	9	22			
-3	12	25	-2	5	10	29-1	11	68
-5	5	0	-3	10	15			
			-5	5	0	30-1	5	0
8-2	5	0						
-3	5	0	19-1	12	17	31-1	9	22
-5	7	0	-2	2	0			
			-3	2	25			
9-1	5	10	-4	4	25			
-2	5	0	-5	7	50			
-3	4	38						
-5	5	0	20-1	2	25			
			-4	10	5			
			-5	11	32			

Table 15. A comparison of some groundwaters to selected water quality criteria. Criteria are for Ontario (M.O.E.) unless otherwise indicated.

Parameter	Water Quality Criteria		Leachate		Background			Other	
	UM28-2	UM29-1	UM28-2	UM29-1	I-UM10-1	II-UM16-3	UM7-1	UM19-1	UM27-6
Cl	250	3185	4362	3185	202	84560	2320	710	46640
SO ₄	250	40	130	40	1360	1080	1680	645	963
F	1.5	1.3	6.6	1.3	0.8	n.d.	0.8	0.36	n.d.
NO ₃	44	1.0	2.0	1.0	< 0.4	< 0.4	0.8	< 0.4	< 0.4
B	1.0	24	120	24	0.8	3.2	7.7	0.73	2.7
Mn	0.05	0.2	0.16	0.2	0.047	1.1	0.03	0.45	1.7
As	0.05	0.003	0.029	0.003	< 0.001	< 0.001	< 0.001	< 0.001	< 0.001
Cd	0.01	< 0.005	< 0.01	< 0.005	< 0.005	< 0.005	< 0.005	0.0008	< 0.005
Cr	0.05	0.043	0.82	0.043	< 0.025	0.042	< 0.025	0.004	0.4
Cu	0.05	0.015	0.15	0.015	0.046	0.46	0.076	0.019	0.26
Pb	0.05	0.032	0.097	0.032	< 0.03	0.086	0.044	0.003	0.11
Se	0.01	0.001	0.004	0.001	< 0.001	< 0.001	< 0.001	< 0.001	< 0.001
Zn	5	0.028	0.47	0.028	0.009	0.036	0.033	0.15	0.25
Benzene	0.01 ^a	0.008	0.027	0.008	0.00001	0.0008	0.004	< 0.001	0.002
Trichloroethylene	0.01 ^a	< 0.0001	< 0.0001	< 0.0001	< 0.0001	n.d.	0.0002	< 0.0001	< 0.0001
Carbon tetrachloride	0.005 ^a	< 0.0001	< 0.0001	< 0.0001	< 0.0001	n.d.	< 0.0001	< 0.0001	< 0.0001
PCBs	0.0001 ^a	0.032	0.032	n.d.	n.d.	n.d.	n.d.	< 0.0002	n.d.
Di-n-butyl phthalate	0.77 ^a	0.086	n.d.	0.086	0.005	n.d.	0.003	< 0.005	0.006
1,4-Dichlorobenzene	0.0047 ^a	0.009	n.d.	0.009	0.0001	n.d.	0.00004	< 0.0001	< 0.0001
Parameters exceeding criteria		9		3	1	5	4	3	7

^aNew York State Standard

n.d. - not determined

Appendix A

PROTOCOLS EMPLOYED FOR CHEMICAL SAMPLING

Appendix C

RESULTS OF ROUTINE CHEMICAL ANALYSES

Appendix B

METHODS FOR THE CHEMICAL ANALYSIS OF GROUNDWATER

SAMPLES

Appendix D

REPORT OF TRACE ORGANIC ANALYSES - MINISTRY OF
THE ENVIRONMENT

Appendix E

REPORT OF TRACE ORGANIC ANALYSES - STANFORD
UNIVERSITY

Appendix F

A PAPER DESCRIBING THE MULTILEVEL MONITORING
DEVICE

UPPER OTTAWA STREET LANDFILL SITE STUDY

20 Jackson Street West, Suite 412, Hamilton, Ontario L8P 1L2 (416) 524-2434

Chairman:

Dr. Arthur N. Bourns

Committee Members:

Dr. Dennis R. McCalla

A. L. (Sandy) McCallion

Dr. James B. Osbaldeston, M.D.

February 27, 1985

Research Director:

Anne Koven

Romeo Palombella
Engineering Dept.
Region of Hamilton-Wentworth
6th Floor
City Hall
71 Main Street West
Hamilton, Ontario

Dear Mr. Palombella:

Recently the Committee became aware that the Hearing on the Proposed Hamilton Transportation Corridor would be discussing the hydrogeological implications of the planned construction of a highway near the east side of the Upper Ottawa Street landfill.

Beginning in April of 1981, the Upper Ottawa Street Landfill Study Committee, which was established by the Ontario Ministry of Health, has been conducting an extensive investigation of the potential health and environmental impacts associated with the site. The Committee anticipates completing a final report of its investigation this summer.

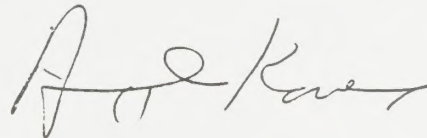
As part of its overall investigation, the Committee commissioned Dr. John Cherry, at the University of Waterloo, to conduct a comprehensive hydrogeological study of the landfill site. Dr. Cherry's work was discussed in the Committee's Interim Report of May 1983 and the Committee received his final report on February 8, 1985.

Upon receiving Dr. Cherry's report, the Committee was of the opinion that it would be irresponsible not to provide this information to the parties at the hearing when one of the issues being discussed is related to the hydrogeological aspects of the landfill site. It is the Committee's view that Dr. Cherry's report will add to existing knowledge on this matter. Therefore, the Committee requested and received approval from the Ministry of Health to release Dr. Cherry's report in advance of the Committee's final report.

I received a letter from Turkstra Partners requesting permission for their client's hydrogeological consultant to discuss with Dr. Cherry the findings of his report, and I understand that Mr. Thomas Lederer, the lawyer for the Region, made a similar request to Dr. Bourns. The Committee has no objection to either their consultant or the Region's hydrogeological consultant discussing the report with Dr. Cherry, however, Dr. Cherry will restrict his comments to answering questions of clarification about the contents of his report. Dr. Cherry is the Committee's consultant, therefore, it would be inappropriate for him to make statements that go beyond the contents of the report he has prepared for the Committee.

Please find enclosed three copies of Dr. Cherry's report. The Committee does not intend to participate in the hearings but hopes that the information contained in Dr. Cherry's report will be useful for all parties involved in the hearings.

Yours sincerely,

A handwritten signature in dark ink, appearing to read "Anne Koven", with a stylized, cursive script.

Anne Koven
Research Director

AK/tlw
Ecl.

c.c. John Cherry, University of Waterloo
Geraldine Lindley, Turkstra Partners
John Mayes, Ministry of the Environment
Dr. Lesbia Smith, Ministry of Health

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